Technical Report

TR-99-28

Deep repository for long-lived low- and intermediate-level waste

Preliminary safety assessment

Svensk Kärnbränslehantering AB

November 1999

Svensk Kärnbränslehantering AB
Swedish Nuclear Fuel
and Waste Management Co
Box 5864
SE-102 40 Stockholm Sweden
Tel 08-459 84 00
+46 8 459 84 00
Fax 08-661 57 19
+46 8 661 57 19

Deep repository for long-lived low- and intermediate-level waste

Preliminary safety assessment

Svensk Kärnbränslehantering AB

November 1999

Keywords: Safety assessment, LILW, Deep repository

Foreword

This report presents a preliminary safety assessment for the deep disposal of Swedish long-lived low and intermediate level waste. The aim of the study is to evaluate the present design and the importance of the site for the safety in a long time perspective. It is the first preliminary safety report made on the total predicted amount of this waste, with all relevant radionuclides included. Previous safety studies have been made on the disposal of parts of the waste or in order to analyse the function of the near-field barriers of a previous repository design.

"Preliminary" in this context means that the scope has been limited and not all aspects have been analysed or all assumptions tested. A repository for this waste will, according to present plans, not be in operation before 2030. This study should be considered as an early step in that direction and the report is meant to be used as a guide for further development and investigations.

The following organisations and persons have been involved in the work and been active in the writing of this report.

Yngve Albinsson	Chalmers University
Bert Allard	University of Örebro
Karin Brodén	Studsvik RadWaste AB
Jan Carlsson	SKB
Stellan Holgerson	Chalmers University
Johan Holmén	Golder Grundteknik AB
Harry Larsson	SKB
Fred Karlsson	SKB (Chairman)
Ulrik Kautsky	SKB
Lena Morén	SKB
Luis Moreno	KTH
Ivars Pavasars	University of Linköping
Michael Pettersson	Kemakta Konsult AB
Kristina Skagius	Kemakta Konsult AB
Marie Wiborgh	Kemakta Konsult AB

The project was managed by SKB and this report was edited by Marie Wiborgh. Also thanks to Richard Nord for translation from Swedish to English.

Abstract

A preliminary safety assessment has been performed of a deep repository for long-lived low- and intermediate-level waste, SFL 3-5. The purpose of the study is to investigate the capacity of the facility to act as a barrier to the release of radionuclides and toxic pollutants, and to shed light on the importance of the location of the repository site. A safety assessment (SR 97) of a deep repository for spent fuel has been carried out at the same time. In SR 97 (SKB, 1999), three hypothetical repository sites have been selected for study. These sites exhibit fairly different conditions in terms of hydrogeology, hydrochemistry and ecosystems. To make use of information and data from the SR 97 study, we have assumed that SFL 3-5 is co-sited with the deep repository for spent fuel. A conceivable alternative is to site SFL 3-5 as a completely separate repository.

The focus of the SFL 3-5 study is a quantitative analysis of the environmental impact for a reference scenario, while other scenarios are discussed and analyzed in more general terms. Migration in the repository's near- and far-field has been taken into account in the reference scenario. Environmental impact on the three sites has also been calculated. The calculations are based on an updated forecast of the waste to be disposed of in SFL 3-5. The forecast includes radionuclide content, toxic metals and other substances that have a bearing on a safety assessment.

The safety assessment shows how important the site is for safety. Two factors stand out as being particularly important: the water flow at the depth in the rock where the repository is built, and the ecosystem in the areas on the ground surface where releases may take place in the future. Another conclusion is that radionuclides that are highly mobile and long-lived, such as ^{36}Cl and ^{93}Mo, are important to taken into consideration. Their being long-lived means that barriers and the ecosystems must be regarded with a very long time horizon.

Sammanfattning

En preliminär säkerhetsanalys har genomförts av ett djupförvar för långlivat låg- och medelaktivt avfall, SFL 3-5. Syftet med studien är att undersöka konstruktionens förmåga att fungera som en barriär mot uttransport av radionuklider och miljöfarliga ämnen, samt att belysa betydelsen av förvarsplatsens läge. Parallellt har en säkerhetsanalys (SR 97) genomförts av ett djupförvar för använt bränsle. I SR 97 studien (SKB, 1999) har man valt att studera tre hypotetiska platser för lokalisering. Dessa platser uppvisar tämligen olika förhållanden vad gäller hydrogeologi, vattenkemi och ekosystem. För att utnyttja information och data från SR 97 studien har vi utgått från att SFL 3-5 samlokaliseras med djupförvaret för använt bränsle. Man kan även tänka sig att lokalisera SFL 3-5 som ett helt fristående förvar.

Tyngdpunkten i SFL 3-5 studien är en kvantitativ analys av omgivningspåverkan för ett referensscenario, medan andra scenarier diskuteras och belyses mer översiktligt. För referensscenariot har migrationen i förvarets närzon och fjärrzon beräknats. Även omgivningspåverkan på de tre platserna har beräknats. Beräkningarna utgår från en uppdaterad prognos på avfallet som ska deponeras i SFL 3-5. I prognosen ingår innehållet av radionuklider, miljöfarliga metaller och andra ämnen av betydelse för en säkerhetsanalys.

Säkerhetsanalysen visar vilken betydelse platsen har för säkerheten. Två saker framstår som speciellt viktiga: vattenflödet på det djup i berget där förvaret förläggs och ekosystemet i de områden på markytan där utsläpp kan ske i framtiden. En annan slutsats är att radionuklider som är mycket rörliga och har lång livslängd, t ex ^{36}Cl och ^{93}Mo, är viktiga att beakta. Den långa livslängden innebär att barriärerna och ekosystemen måste betraktas i ett mycket långt tidsperspektiv.

Executive Summary

Background

Low- and intermediate-level waste (LILW) from the operation of the nuclear power plants is treated, packaged and shipped to the final repository SFR in Forsmark. To this quantity is added a smaller quantity of LILW from research, industry and medicine that is packaged and temporarily stored in Studsvik, which also produces similar waste. Studsvik has a rock cavern for interim storage.

Some of the waste in Studsvik contains too much long-lived material to be received at SFR, however. This is primarily waste from research, and such waste is currently being set aside. Some spent internal components from the nuclear power reactors, including core components, also contain too much long-lived radionuclides to be able to be sent to SFR. They are stored in CLAB or at the nuclear power plants.

A repository for long-lived LILW must also be able to receive the short-lived waste from the operation of CLAB and the encapsulation plant, such as spent filter resins arising after SFR has been closed and sealed. Added to this is the decommissioning waste from CLAB and the encapsulation plant, as well as various leftover equipment in the form of transport casks/containers and the like. Operational waste and decommissioning waste make up roughly half of the volume of the total inventory.

Repository

The plans call for long-lived LILW to be disposed of in a deep repository, SFL 3-5. All waste will be treated and packaged before it is shipped to the deep repository. No liquid waste may be sent there without first having been solidified and stabilized. This is the same rule that applies to disposal in SFR.

A layout has been devised for a deep repository for long-lived LILW. This design has now been evaluated by means of a preliminary safety assessment, which is the first time such an assessment has been carried out for this waste. The proposed design is largely based on experience from construction and operation of the BMA rock vault in SFR-1. The long-lived waste is placed in enclosures of concrete inside a rock cavern. Two such rock caverns are needed, SFL 3 and SFL 5, see Figure 1. Some waste that arises at a late stage and is not long-lived is placed in the transport tunnels surrounding the two rock vaults. The part of the transport tunnels where waste is emplaced is designated SFL 4, see Figure 1.

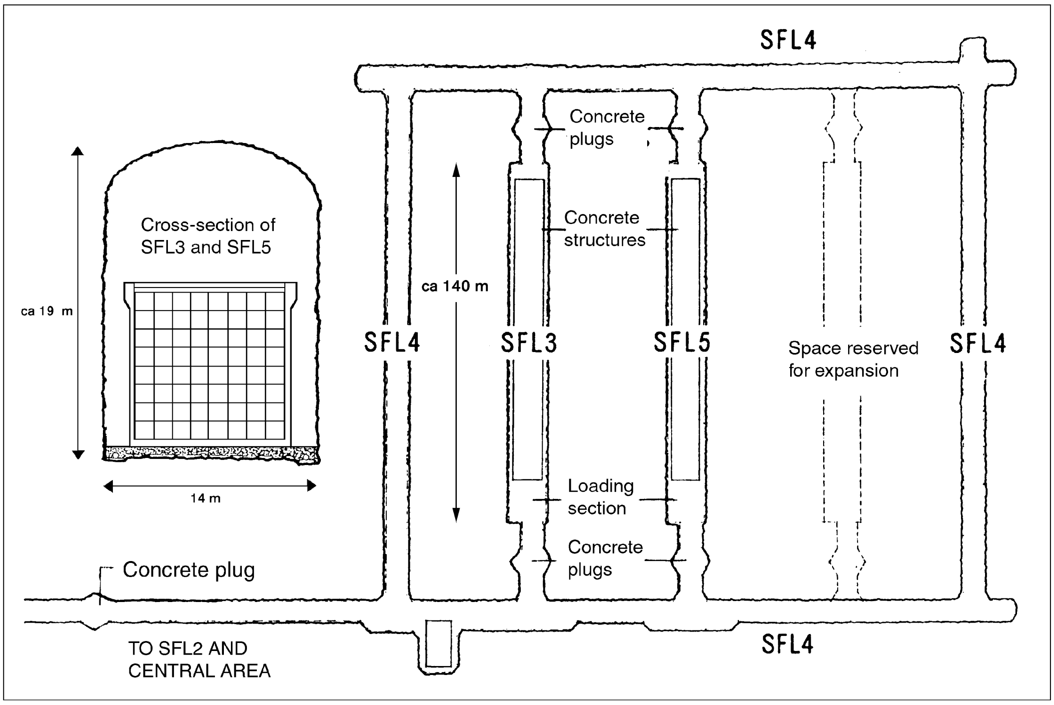

Figure 1Plan of rock caverns in SFL 3-5.

Harmful effects on the biosphere caused by radioactive substances and (non-radioactive) toxic pollutants are prevented by packaging the waste in containers of steel and concrete. In the repository, the containers in SFL 3 and SFL 5 are surrounded by concrete walls and by backfill – concrete mortar is used inside the concrete walls of the enclosure, while crushed rock is used outside. Only less active waste packages are placed in tunnels with crushed rock as the only backfill. The concrete and the rock are the most important barriers to the escape of radionuclides and toxic pollutants. The concrete contains the radioactive and chemotoxic substances, prevents (together with outer draining backfill) water flow through the waste and sorbs dissolved radionuclides and toxic pollutants. The rock protects the repository while limiting the flow of water and the transport of radionuclides and toxic pollutants.

This repository can be built as a part of the deep repository for spent fuel or completely separately. In this assessment we have chosen to make SFL 3-5 a part of the deep repository for spent fuel. Since the repositories contain different materials, for example copper and bentonite in the deep repository for spent fuel compared with concrete and organic material in SFL 3-5, they are kept separate. This is to prevent the different materials from interacting with each other. A distance of 1 km has been chosen as the minimum distance. In this study it is assumed that SFL 3-5 is located at a depth of 300 m or deeper.

Waste

Some of the long-lived LILW already exists, for example waste from research with long-lived radionuclides. Such waste is treated, packaged and temporarily stored in Studsvik. However, much of the long-lived LILW will not arise until later in the future. This includes the reactor internals, which are currently installed in power reactors. The short-lived operational waste and the decommissioning waste from CLAB and the encapsulation plant, which will arise in the future when SFR has been closed, must also go to the deep repository.

It is above all three kinds of information that are needed about the waste: volume of waste, content of different radionuclides and content of different materials and substances of importance to safety. Waste that is already packaged is available from previous research. Its volume is naturally known, and we have a good idea about what inactive substances are contained in the waste. The radionuclide content is also recorded. What needs to be calculated is the quantity of the difficult-to-measure radionuclides, which was not originally known. ^{36}Cl, formed by neutron activation of chlorine impurities in steel, is an example of such a long-lived radionuclide. We have calculated both waste quantities and content of radionuclides and other substances of interest for the safety assessment for the waste that has not yet been produced. According to present-day forecasts, the waste to be disposed of in SFL 3-5 has a volume of about 25,000 m^3 in the repository. A compilation of volumes, activity and radiotoxicity for the waste in the different parts of the repository is shown in Table 1.

Table 1 Information on the waste to be disposed of in SFL 3-5.

Characteristics	**SFL 3**	**SFL 4**	**SFL 5**
Total waste volume	5,600 m^3	10,000 m^3	9,700 m^3
Origin and volume of waste	LILW[a)] from Studsvik, 1,800 m^3 Operational waste from CLAB[b)] and Encapsulation plant, 3,800 m^3	Decommissioning waste from CLAB and Encapsulation plant, 2,200 m^3 Storage canisters from CLAB, 6,500 m^3 Transport casks and transport containers, 1,300 m^3	Core components and reactor internals, BWRs[c)], 7,800 m^3 PWRs[d)], 1,800 m^3 Decommissioning waste from Studsvik, 50 m^3
Packaging	Concrete moulds and steel drums	Steel containers	Long concrete moulds with inner steel cassette
Activity content in 2040	$2 \cdot 10^{16}$ Bq	$7 \cdot 10^{13}$ Bq	$1 \cdot 10^{17}$ Bq
Dominant radionuclides (activity, Bq)	Ni-63, Co-60, Ni-59, C-14, Cs-137	Co-60, Fe-55, Ni-63, Sb-125, Cs-137	Ni-63, Co-60, H-3, Ni-59, Fe-55
Dominant radionuclides, (radiotoxicity, Sv)	Co-60, Ni-63, Am-241, Pu-239, Pu-240	Co-60, Cs-134, Cs-137, Fe-55, Sr-90	Co-60, Ni-63, Fe-55, C-14, Ni-59

[a)] Low- and intermediate-level waste
[b)] Central Interim Storage Facility for Spent Nuclear Fuel
[c)] Boiling Water Reactor
[d)] Pressurized Water Reactor

Safety assessment

A preliminary safety assessment has been carried out in which the potential dispersal and environmental impact of the radionuclides is calculated. Chemotoxic pollutants such as lead, beryllium and cadmium that are present in the repository have also been included in the calculations. The assessment is based on the assumption that SFL 3-5 is located on the same site as the deep repository for spent fuel. This has enabled us to choose the same premises as for the more comprehensive and thorough safety assessment of the repository for spent fuel, and utilize some of the same results. The safety assessment for the repository for spent fuel is designated SR 97 and is being conducted in parallel with the assessment of SFL 3-5 (SKB, 1999). The three hypothetical sites included in SR 97 are called Aberg (Äspö), Beberg (Finnsjön) and Ceberg (Gideå). Naturally, full freedom of choice will exist in the future in the siting of a repository in Sweden as far as the positioning of both the deep repository for spent fuel and the repository for long-lived LILW is concerned. Other alternatives are also possible, for example locating SFL 3-5 at SFR or building it as a completely separate repository. This does not affect the fundamental reasoning and conclusions presented in this report.

Reference scenario

A reference scenario has been formulated which depicts the expected evolution of the repository's near field. A stable far field is an important prerequisite for the reference scenario, i.e. that no decisive changes occur in the thermal, hydrological, mechanical and chemical conditions in the rock surrounding the repository. The reference scenario has been used to carry out a quantitative analysis of the repository's performance, i.e. calculations of releases and environmental impact.

The heat generated by radioactive decay in the waste results in an insignificant temperature rise less than 5°C. This occurs during the first 100 years after closure. Water from the surrounding rock will fill the repository after closure, and eventually the water flows and pressure gradients will become steady-state. This may take from a decade or so up to several hundred years, depending on local hydrological conditions. Air trapped in the repository on closure will dissolve in the water and the oxygen will be consumed by e.g. corrosion of steel. The water composition inside SFL 3 and SFL 5 will be affected by the concrete. Most important is the influence on pH, which will exceed 12, at least inside the concrete walls.

In the long time perspective it is the permeability of the near-field barriers to water and gas and the composition of the water in the repository that will be of importance for the liberation and release of both radionuclides and toxic pollutants from the near field. The main processes that will affect the long-term properties of the barriers and the composition of the water have been judged to be:

- Corrosion of steel and aluminium (formation of gas and corrosion products).
- Microbial degradation of organic material in the waste in SFL 3.
- Build-up of gas pressure in the waste packages and concrete enclosure.
- Leaching of cement and concrete (high pH).
- Reactions between leaching products from cement and surrounding gravel fill.

- Precipitation of calcite and brucite (i.e. calcium carbonate and magnesium hydroxide).
- Alkaline degradation of cellulose in the waste in SFL 3 (formation of complexing agents)

In order for radionuclides to be liberated and transported from the near field, they have to dissolve in water or occur as gas. Of the radionuclides that are present in the waste, only organic ^{14}C is expected to be able to be emitted and transported out of the near field as a gas, e.g. if ^{14}C forms methane and mixes with gas from metal corrosion or organic degradation. In the safety assessment we assume that this transport occurs as soon as transport pathways for gas have been created in the near-field barriers (principally concrete walls) and without any retardation whatsoever in them.

The waste to SFL 5, which consists of internal metal components from the reactors, primarily contains induced activity. These radionuclides are not immediately accessible for dissolution in the water, since they are present inside the metal components. The metal has to corrode to render the radionuclides accessible. Since the metal contains both stable and radioactive isotopes of the same elements, e.g. nickel in stainless steel and zirconium in Zircaloy, isotope dilution will reduce the solubility of the radioactive isotope in the water.

Radionuclides, as well as toxic pollutants, that have dissolved in the water in the waste packages can be transported out through the near-field barriers by diffusion and by water flowing through the barriers. This out-transport is retarded by sorption in cement, concrete and gravel fill. Sorption is controlled principally by the composition of the water, where pH, Eh and the presence of organic complexing agents are important parameters. Colloids have not been taken into account in this study. Due to the strong mineralisation of the water in the concrete, no significant colloid concentrations are expected.

Hydrology calculations

The regional hydrology models that have been developed within SR 97 have been used to calculate the specific water flow in the rock around SFL 3-5 on the three sites. The calculations also give the direction of the water flow and travel times for the water from the repository up to the ground surface, as well as where the discharge areas are located. The water flow through the different barriers in the near field has been calculated with a generic hydrological model.

The results of the calculations with the regional hydrology models have not been used directly, but for the sake of comparison have been rounded off to representative water flows. This water flow in Beberg has been assumed to be 1 litre/m^{2}year, in Aberg a factor of 10 higher and in Ceberg a factor of 10 lower. The chosen values differ somewhat from the calculated ones, but the purpose is not to determine the suitability of the sites but above all to ascertain the importance of the water flow. The direction of the water flow in the surrounding rock is also chosen on the basis of the calculation. A horizontal groundwater flow in the rock has been assumed for all three sites. The representative water flows in the rock, together with the near-field calculations, gave a specific water flow of 0.01 litre/m^{2}year in the concrete enclosure and its interior in

SFL 3 and SLF 5, and about 30 litre/m^{2}year in the gravel fill outside when the repository is located in Beberg. The specific water flow in the SFL 4 tunnel has been calculated to be 380 litre/m^{2}year for Beberg. For a repository located in Aberg, all water flows are 10 times higher, and for a repository located in Ceberg 10 times lower.

The flow paths from the repository discharge into areas which, according to the biosphere descriptions in SR 97, are classified as “Archipelago” and “Coast” in Aberg, “Agricultural land” in Beberg and “Peatland” in Ceberg.

Transport of radionuclides and toxic pollutants

The transport of radionuclides and toxic pollutants through the engineered barriers in the repository and out through the rock to the ground surface has been calculated. The expected evolution of the repository’s near-field has been taken into account via the assumptions and choice of data made in the calculations. These assumptions and choices are summarized below.

- The time it takes to refill the repository with water after closure is neglected and water-saturated conditions are assumed to exist at the time of closure, in 2040.

- The radionuclides present in the form of inducted activity in metal components in the waste in SFL 5 are assumed to be liberated as the metal components corrode. The rest of the radionuclides, as well as the toxic pollutants, are assumed to be immediately accessible for dissolution in the water regardless of where they are located in SLF 3-5. Sorption on cement and concrete in the waste packages is taken into account. Similarly, solubility limitations are taken into account, but no further reduction in solubility due to isotope dilution has been included this time.

- Dissolved radionuclides and toxic pollutants are transported out through the waste packages, the concrete barriers and the gravel fill to the surrounding rock, by diffusion and by water flowing through the repository. Sorption in concrete and gravel fill retard the out-transport.

- The water flows in the different repository parts are assumed to be horizontal. Any influence of gas on the direction and size of the water flow is expected to be negligible.

- Sorption and solubility data are chosen based on expected water composition. In waste packages and enclosures, data are chosen for high pH (≥ 12.5), reducing conditions and sodium and potassium concentrations that correspond to a saline penetrating groundwater. The assumption of high salinity in packages and enclosures is conservative for ion-exchange-sensitive nuclides such as caesium. In the gravel fill, on the other hand, sorption data are chosen according to the groundwater on the site in question. The same sorption data are used in the rock in the far field (also used in SR 97).

- The waste in SFL 3 contains cellulose, which can be degraded at high pH. The degradation products include the strong complexing agent isosaccharinic acid, ISA. At worst, ISA could increase the solubility and reduce the sorption of important radioactive substances in SFL 3, such as plutonium. However, calculations show that the concentration of ISA in the waste packages in SFL 3 will be so low that its influence on sorption is negligible. The main reason for this is that ISA itself sorbs on the cement. An increased solubility due to ISA in SFL 3 can, on the other hand, not be ruled out. Such an effect of ISA is therefore included as a variation case in the calculations.

- Mass transport in the far field is assumed to take place with flowing groundwater. The travel time for the water from the repository to the ground surface has been obtained from the calculations with the regional hydrology models. Transport of radionuclides and toxic pollutants is retarded in the rock due to diffusion and sorption in the rock matrix. Sorption and diffusion data are chosen with respect to the groundwater. They are the same values as those chosen for SR 97 as "best estimates" for saline and fresh groundwater. The choices of sorption and diffusion constants are based on experimental results.

- The calculated release of radionuclides to the biosphere is used in turn to calculate a dose to humans in the area. The different ecosystems in the area that are affected by the releases are "Coast" and "Archipelago" in Aberg, "Agricultural land" in Beberg and "Peatland" in Ceberg. We use the same dose conversion factors for the different ecosystems as SR 97. The release of toxic pollutants from the far field to the biosphere has been converted to concentrations in the release areas.

The results of the calculations show that:

- In Aberg, the highest dose is obtained from the release of radionuclides to the "Archipelago" area. The dose does not exceed $4 \cdot 10^{-3}$ µSv/year. The biggest dose contribution is obtained from inorganic ^{14}C in SFL 3 and SFL 5 some 10,000 years after closure.

- In Beberg, the release of radionuclides to "Agricultural land" gives a maximum dose of 3 µSv/year. This occurs 8,000 years after closure and the dose is dominated by ^{93}Mo from SFL 5 and SFL 3.

- In Ceberg, the release of radionuclides to "Peatland" gives rise to a maximum dose of less than 10 µSv/year. This occurs more than 30,000 years after closure of the repository, and the dose is dominated by ^{36}Cl from SFL 5 and SFL 3.

- At all sites releases of the toxic pollutants lead, beryllium and cadmium give rise to concentrations in the ecosystems that are very low compared with measured natural concentrations.

Releases of gaseous radionuclides

The waste in SFL 3-5 is not gaseous. However, the possibility cannot be excluded that organic ^{14}C will eventually be transformed into e.g. methane. In this way, ^{14}C could accompany inactive gas from the repository, for example gas from corrosion of steel and aluminium. Organic ^{14}C is only contained in the waste to SFL 3, and the total inventory has been estimated at less than 10^5 Bq. The collective dose caused by a pulse release of ^{14}C from a deep repository was calculated in SR 95. With the same way of calculating, SFL 3 could give rise to a maximum total collective dose of about 0.04 μmanSv locally and regionally. Since the collective doses to the local and regional population are distributed among a large number of individuals, the annual individual dose will be considerably lower.

Alternative scenarios

Besides the reference scenario, a number of alternative scenarios have been studied. Climate change, seismic activity and human actions are examples of courses of events and activities that could influence the future performance and safety of the repository. In this study, however, environmental impact has only been calculated for the alternative scenario with future drilled wells in the vicinity of the repository.

According to the regional hydrology models, there will not be any releases from SFL 3-5 in any of the wells that exist today. The calculation of environmental impact is therefore based on the assumption that a new well is drilled in a discharge area of the repository. All releases from the repository are transferred to this well, which is assumed to have a capacity representative of the area. The dose to man is calculated using the dose conversion factors obtained for wells in SR 97. The water in the well is used as drinking water by humans and cattle and for irrigation of a small vegetable garden. The exposure pathways are intake of contaminated well water, vegetables irrigated with this water and intake of milk and meat from cattle that have drunk the water.

The consequence of the toxic pollutants lead, beryllium and cadmium are reported only as releases in the well. The calculated metal concentrations are compared with applicable guideline values for drinking water.

The release of radionuclides from SFL 3-5 to a future well in Beberg gives rise to a dose of 2 μSv/year and in Ceberg to a dose of 0.3 μSv/year. The highest doses occur in Aberg, the site with the highest groundwater flow. The maximum dose for release from SFL 3-5 to a well in Aberg is 25 μSv/year. The dose is completely dominated by the release of ^{93}Mo and the contribution from SFL 5 accounts for approximately 90% of this.

A future drilled well in the vicinity of the repository – regardless of whether the repository is sited in Aberg, Beberg or Ceberg – results in concentrations of lead, beryllium and cadmium in the well water that are well below applicable guideline values for drinking water.

Discussions and conclusions

A comparison between the three hypothetical sites shows that both geohydrological and biosphere conditions are of importance for the calculated impact on the environment. The ecosystem in the release area is of particularly great importance for the dose obtained from the release of radionuclides from the far field. Releases to “Coast” or “Archipelago” give much lower doses than releases to “Agricultural land” or “Peatland”. The residence time of the groundwater in the rock and the composition of this water are of less importance, since the radionuclides that give the greatest calculated release and the highest dose are low-sorbing and long-lived (^{36}Cl and ^{93}Mo). The size of the water flow in the rock surrounding the repository makes a difference, however, since it influences transport in the near-field barriers. A high water flow, such as in Aberg, leads to a higher release from the near field than a low water flow, such as in Ceberg. Although diffusion in the concrete barriers does limit the transport out from the near-field barriers in SFL 3 and SFL 5, the resistance to diffusion is not great enough to compensate for the disadvantages of a high water flow in the environment, such as in Aberg.

For high water flows, such as in Aberg, the barriers in SFL 3 and SFL 5 need to be improved. The calculations show that the existing design can give rise to excessively high releases of long-lived, low-sorbing radionuclides from the near field. Various possible improvements have been discussed. One possibility is to increase the thickness of the walls, floor and roof of the enclosure and thereby increase the diffusion resistance. But this requires drastic increases, for even if the thickness is doubled the release to surrounding rock is only reduced by about a factor of 2. Another way to limit diffusion is to reduce the surface area through which diffusion takes place. This can be achieved by the use of impervious (diffusion-proof) materials around the waste. The difficulty lies in finding impervious materials that can be used and will last for the very long times required.

A third possibility is to backfill the void outside the enclosure in SFL 3 and SFL 5 with a material that is less permeable to groundwater than a gravel fill. But for this to result in a lower release of radionuclides from the near field, the hydraulic conductivity in the enclosure and its interior, as well as in the backfill, must be equal to or lower than the hydraulic conductivity in the surrounding rock. The backfill must not be too impervious, however. Gas formed in the repository should preferably be able to migrate out without building up a pressure that displaces water from the enclosure. If a backfill exists that can meet these conditions and last for a very long time, it could reduce the release of long-lived low-sorbing radionuclides up to a factor of 10.

Whether or not a site is good enough for the chosen design can only be determined after geoscientific investigations and characterization of the ecosystems on the site. Of particular importance are hydrogeological surveys, which include measurements in boreholes down to repository depth. The water chemistry, e.g. the salinity of the groundwater, and the ease with which the rock caverns can be built are naturally also of importance, to mention just two of the factors included in a complete site characterization.

Methods and technology exist for carrying out site investigations. Calculations of water flows and radionuclide releases to the conceivable ecosystems will then show whether the design and the site are possible, i.e. whether they allow safe disposal of long-lived low- and intermediate-level waste.

The principal conclusions of the study are:

- The radionuclides in the waste that are the most important for the evaluation of safety are those that are highly mobile and have a long life. The long life means that the barriers and the ecosystems must be regarded in a very long time perspective.
- To limit uncertainties in calculated environmental consequences, efforts should be concentrated on limiting the uncertainties for the radionuclides that are predominant for the dose, ^{36}Cl and ^{93}Mo. In addition, studies leading to a greater understanding of radionuclide availability in the waste, migration in the barriers and dose effects are of importance.
- The properties of the site are of importance for safety. Two aspects stand out as being particularly important: the water flow at the depth in the rock where the repository is built, and the ecosystem in the areas on the ground surface where releases may take place in the future.
- An unfavourably high water flow in the rock around the repository can be compensated for by better barriers in the near field. Satisfactory function must however be sustained over a very long time span. This requires materials that are durable in the chemical and mechanical environment of the repository.

Contents

List of Abbreviations

List of figures

List of tables

1 Introduction

1.1 Background

Some of Studsvik's low- and intermediate-level waste (LILW) and some internal components from the power reactors contain too much long-lived radionuclides to be disposed of in SFR. The long-lived waste needs to be isolated much longer than the ordinary operational waste that goes to SFR. The rock in a deeper repository provides much better protection against changes in the very long term. SKB is therefore considering disposing of all long-lived waste, including the spent fuel, in a deep repository (SKB, 1998). The deposited LILW will in this case be kept well-separated from the canisters with the spent fuel (about 1 km) so that the different types of waste do not affect each other. Other alternatives are also conceivable, such as locating the deep repository for long-lived LILW at SFR or building it as a completely separate repository. This does not affect the fundamental reasoning and conclusions in this report, provided that the repository is built as a deep repository. In this study we assume that the repository for long-lived LILW is located at a depth of 300 m or more.

According to existing plans, a repository for LILW will be in operation at roughly the same time as the deep repository for spent fuel (after the initial phase with demonstration deposition of canisters). Since CLAB and the encapsulation plant must be in operation during this period, some ordinary short-lived operational waste will arise. If SFR has already been closed when this occurs, the ordinary operational waste from CLAB and the encapsulation plant will be disposed of in the deep repository together with the long-lived LILW.

A first draft of the design of a deep repository for long-lived LILW, SFL 3-5, is described in the SKB report PLAN 82 (1982), accompanied by a more detailed design in PLAN 93 (1993). Based on the latter, a pre-study was carried out in 1992–1995 for the purpose of determining how well the repository and the near field performed as barriers to the release of radionuclides (Wiborgh, 1995). The analysis also included some chemotoxic elements that may be present in the waste, for example the metals lead and beryllium. Based on the results, a new and simplified design of SFL 3-5 was proposed. This was the first feasibility study, and it contained an initial inventory of the waste, both existing waste and waste expected in the future. This inventory has now been updated and a new compilation exists of the waste, what materials it consists of and what radionuclides and other substances it contains that may be of importance in a safety assessment (Lindgren *et al.,* 1998).

1.2 Purpose and outline of the study

Based on the new compilation of the waste and the new design, a safety assessment has been carried out. The purpose of the present report is to evaluate the design and the general importance of the choice of site for safety with the aid of this safety assessment. The safety assessment is only preliminary, but it is the first time a more or less complete assessment has been conducted of long-term safety. The preceding assessment of repository performance covered only a limited number of radionuclides, and only releases from the repository barriers were calculated (Wiborgh, 1995). The new preliminary safety assessment presented here covers virtually all expected nuclides, and the release is calculated all the way out into the biosphere, including dose to individuals.

In parallel with the safety assessment of SFL 3-5, a much more comprehensive and thorough assessment has been performed of a deep repository for spent fuel. The new safety assessment for the deep repository for spent fuel is designated SR 97 (SKB, 1999). So that we can make the greatest possible use of the premises and results from SR 97, we have chosen to locate SFL 3-5 about 1 km from the deep repository for spent fuel (which is not really necessary, SFL 3-5 can naturally be located completely separate from this repository).

The distance of about 1 km between the deep repository for spent fuel and SFL 3-5 has given rise to some differences in the use of site-specific data in the assessment. For example, the estimated emission points to the biosphere are not the same for SFL 3-5 and the deep repository for spent fuel. Furthermore, we have not considered it warranted to carry out the same detailed hydrogeological calculations in the far field for SFL 3-5 as for the deep repository for spent fuel, since the objective is a preliminary safety assessment that will guide the continued work.

Three hypothetical sites named Aberg, Beberg and Ceberg have been chosen for the safety assessment of the deep repository for spent fuel. These sites are modelled on the three different sites in Sweden where SKB has performed geological, hydrogeological and hydrogeochemical investigations – Äspö, Finnsjön and Gideå – and the geoscientific data for Aberg, Beberg and Ceberg are taken from these sites (Walker *et al.*, 1997 and Laaksoharju *et al.*, 1998). We have chosen to use more or less the same data for the assessment of SFL 3-5.

The three hypothetical sites exhibit fairly different hydrogeological and hydrochemical conditions. The biotopes are also quite different inasmuch as Aberg is located near the coast, Beberg has agriculture and forests, while Ceberg has forests and peat bogs. The different conditions on the sites lead to somewhat different results in the safety assessment. This is an advantage in this context, since it shows what the choice of site means for SFL 3-5.

Figure 1-1 provides a general description of the scope and outline of the safety assessment. The focus of the study is a quantitative analysis of the environmental impact for a reference scenario, while other scenarios are discussed and analyzed on a more general level. The assessment also includes a simple estimate of dose from releases of radionuclides in the gas phase, plus an assessment of the consequences of a change in the repository design.

The quantitative analysis of the reference scenario includes calculations of the migration of radionuclides and some toxic metals in the repository's near and far fields for a defined content of radionuclides and toxic metals in the waste, a reference inventory. This reference inventory has been arrived at by making an inventory and characterization of the expected waste to be sent to SFL 3-5. Assumptions and data that enter into the calculations of migration in the near field have been chosen on the basis of an analysis of the expected evolution of the near field and results from experimental research programmes being pursued at the Chalmers University of Technology (cement and radionuclides) and at the Universities of Linköping and Örebro (organic matter and colloids). The numerical model used for the calculations of migration in the near field is a further development of the near-field model used for the deep repository for spent fuel in SR 97. The near-field model for SFL 3-5 can, in contrast to the near-field model for the repository for spent fuel, handle both diffusive and advective transport in the near-field barriers. For the calculations of migration in the far field, we have used the same numerical model and, as far as possible, the same data as in the analysis of the deep repository for spent fuel in SR 97.

The water flows in the near-field barriers have been chosen on the basis of the results of calculations with a generic model of the hydrology in the near field and the results of calculations with the regional hydrology models used in SR 97. The generic near-field model gives water flows in the near-field barriers as a function of the size and direction of the groundwater flow in the surrounding rock. The regional hydrology models give the prerequisites, size and direction of the groundwater flow in the surrounding rock, plus information on travel times for the groundwater from the repository to the ground surface. The latter is used to calculate migration in the far field. The calculations with the regional hydrology models also show where the discharge areas on the three sites are located.

Environmental impact, in the form of dose from radionuclides and concentration in release areas for toxic metals, has then been calculated using information and data from the biosphere studies in SR 97 regarding the ecosystems present on the three sites.

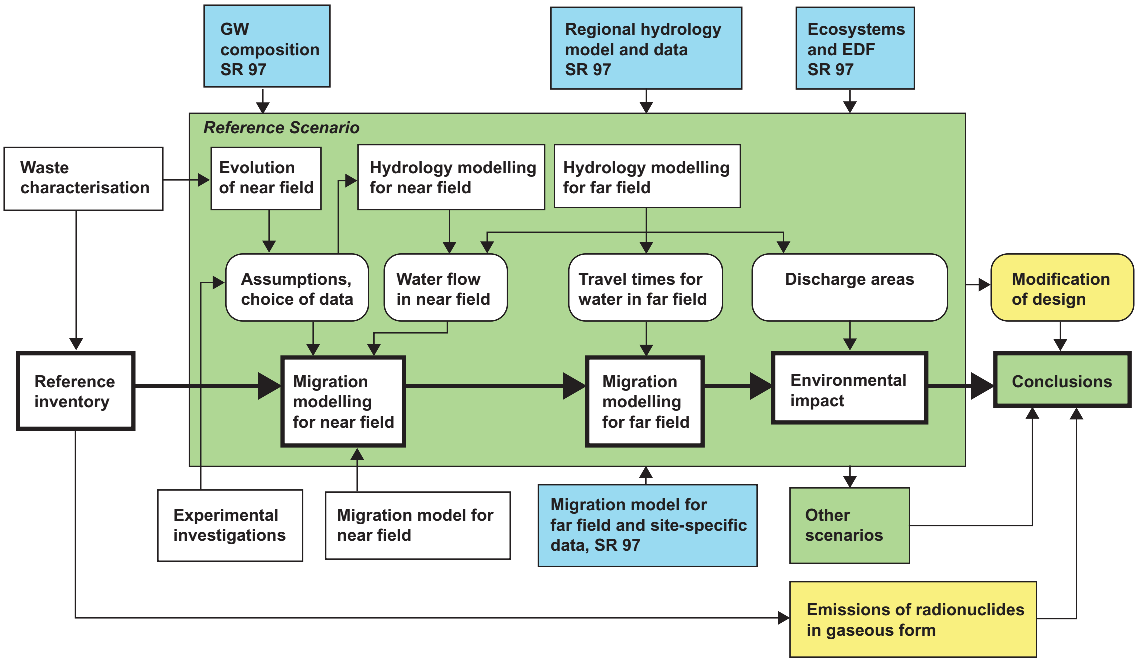

Figure 1-1 Information flow diagram for the preliminary safety assessment of SFL 3-5.

1.3 Outline of the report

Chapter 2 describes the waste and its content of radionuclides. The design of different waste packages is presented. Materials of importance for the long-term properties of the waste packages are described, such as different metals, cement and organic materials. Chemotoxic pollutants contained in the waste are also included in the compilation. The total waste volume has been estimated at 25,000 m^3.

The deep repository for LILW consists of three different repository parts: SFL 3, SFL 4 and SFL 5. The repository and the design of the repository parts are presented in Chapter 3. There are great similarities to the rock chambers in SFR, and the waste will also be treated and packaged in the same or similar ways as in SFR. This chapter describes the materials used in the repository's barriers and their properties, as well as conditions in the repository during construction and operation. The reference scenario in Chapter 6 describes how the barriers will subsequently evolve with time.

Chapter 4 briefly describes the properties of the geosphere on the three hypothetical repository sites Aberg, Beberg and Ceberg and the positioning of SFL 3-5 on the different sites.

Chapter 5 describes in general terms how the calculations for the biosphere have been carried out.

The expected evolution of the repository's near field for a reference scenario is described in Chapter 6. This description serves as a basis for the quantitative analysis.

Chapter 7 discusses the hydraulic properties of the repository and the hydrology in the near field (the host rock surrounding the facility), and presents the hydrology calculations that have been used as input data to the radionuclide transport calculations.

Chapter 8 presents the results of radionuclide transport calculations performed for the reference scenario.

Chapter 9 describes some other scenarios and presents the assessment of the environmental impact of a future drilled well in the vicinity of the repository.

Finally, the results of the safety assessment and the conclusions are summarized and discussed in Chapter 10.

1.4 References

Laaksoharju M, Gurban I, Skårman Ch, 1998. Summary of hydrochemical conditions at Aberg, Beberg and Ceberg. SKB Technical Report TR 98-03, Svensk Kärnbränslehantering AB, Stockholm.

Lindgren M, Pers K, Skagius K, Wiborgh M, Brodén K, Carlsson J, Riggare P, Skogsberg M, 1998. Low and intermediate level waste in SFL 3-5: Reference inventory. A compilation of low and intermediate level waste to be sent to SFL 3-5. Reg. No 19.41/DL31, Svensk Kärnbränslehantering AB, Stockholm.

PLAN 82, 1982. Radioactive waste management plan. Plan 82. Part 1: General. SKBF/KBS Technical Report TR 82-09:1, Svensk Kärnbränslehantering AB, Stockholm.

PLAN 93, 1993. Plan 93. Costs for management of radioactive waste from nuclear power production. SKB Technical Report TR 93-28, Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1998. Systemredovisning av djupförvaring enligt KBS-3-metoden. SKB report R-98-10, Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1999. SR 97 – Main Report. Svensk Kärnbränslehantering AB, Stockholm.

Walker D, Rhén I, Gurban I, 1997. Summary of hydrogeologic conditions at Aberg, Beberg and Ceberg. SKB Technical Report TR 97-23, Svensk Kärnbränslehantering AB, Stockholm.

Wiborgh M, (ed.) 1995. Prestudy of final disposal of long-lived low and intermediate level waste. SKB Technical Report TR 95-03, Svensk Kärnbränslehantering AB, Stockholm.

2 Long-lived low- and intermediate-level waste

2.1 Introduction

The waste that will be disposed of in SFL 3-5 is described in this chapter. Much of the waste will not arise until some time in the future, for example the internal components currently installed in power reactors. The short-lived operational waste and the decommissioning waste from CLAB and the encapsulation plant will also arise in the future. The waste that already exists today includes waste from previous research on long-lived nuclides. Such long-lived low- and intermediate-level waste is treated, packaged and interim-stored at Studsvik. To carry out a safety assessment, we must therefore estimate both existing and planned waste. The collected information on the waste that is used in the safety assessment is designated reference inventory. The reference inventory in this report has been arrived at by revising the waste characterization from the previous feasibility study (Lindgren *et al.*, 1994). The revision has not entailed any major changes as regards waste volumes and material quantities. The biggest difference is that many more radionuclides have been included this time. A more or less complete radionuclide inventory has been calculated, which has been made possible by the use of correlation factors (Lindgren *et al.*, 1998). The total activity content in each repository part has not been changed so much due to this, however, but is of the same order of magnitude as in the previous waste inventory. The reason is of course that the dominant radionuclides were already included in the previous inventory.

The origin of the waste is described below, where forecasts are made of waste volumes, material quantities and activity contents in all parts of SFL 3-5. The waste in the different repository parts is then described in greater detail. Finally, the uncertainties in estimates of quantities, volumes and activity contents are discussed.

2.2 Waste and packagings in SFL 3-5

2.2.1 SFL 3

Long-lived low- and intermediate-level waste (LILW) which is produced, collected and stored at Studsvik will be ultimately disposed of in SFL 3. This also applies to other LILW from Studsvik, if it for any reason fails to fulfil the criteria for final disposal in SFR. Such waste will also be sent to SFL 3. Furthermore, all low- and intermediate-level operational waste produced at CLAB and the encapsulation plant after closure of SFR 1 (no earlier than 2012) will be emplaced in SFL 3. Standard concrete moulds or 200-litre steel drums will mainly be used for disposal in SFL 3.

Studsvik's own research activities give rise to radioactive waste. Moreover, Studsvik collects radioactive waste products from the National Defence Research Establishment, FOA, as well as from industry, medical care, universities and colleges. The waste contains e.g. precipitates, filters, replaced components from the research reactor at Studsvik, radiation protection equipment, glove boxes and radiation sources.

Low- and intermediate-level operational waste from CLAB will be shipped to SFR 1 for final disposal as long as the facility is in operation, but waste that arises afterwards will be disposed of in SFL 3. The encapsulation plant is planned to be in full operation from 2020 until 2040, and all operational waste from there will be stored in SFL 3. The waste consists of ion exchange resins and filter aids from clean-up of the storage pools for spent fuel in CLAB and similar systems in the encapsulation plant. Furthermore, small quantities of solid waste are generated by service and maintenance activities.

LILW from Studsvik is mainly placed in 200-litre steel drums or in reinforced concrete moulds, see Figure 2-1. The waste from CLAB and the encapsulation plant is placed in concrete moulds with a wall thickness of 10 cm and the same outside dimensions as Studsvik's concrete moulds (1.2 × 1.2 × 1.2 m).

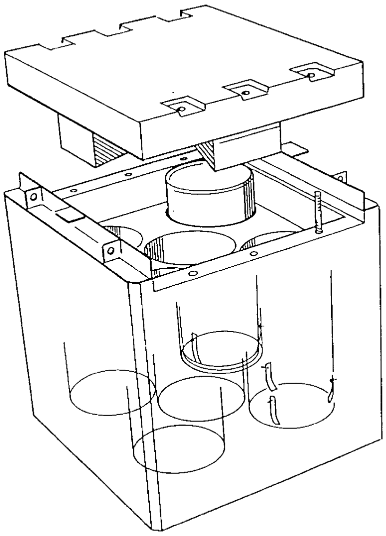

Figure 2-1 Concrete moulds (1.2 × 1.2 × 1.2 m) with five holes for 80-litre steel drums.

Special concrete boxes are used for plutonium-contaminated glove boxes from Studsvik, and smaller quantities of waste from dismantling of the R1 reactor are packaged in steel cases called Berglöf boxes (1.2 × 0.8 × 0.8 m).

2.2.2 SFL 4

SFL 4 consists of the tunnel system outside SFL 3 and SFL 5, where low-level waste will be deposited at a late stage. The waste consists of decommissioning waste from CLAB, decommissioning waste from the encapsulation plant, storage canisters for spent fuel from CLAB, cylindrical transport casks for spent fuel and core components, and radiation-shielded transport containers for ILW.

The decommissioning waste from CLAB consists of metal components and concrete that has been contaminated after a long period of operation. Contaminated metal components will be found in the pools' cooling and clean-up systems. The concrete is chipped off of contaminated surfaces in the buildings during decommissioning. The storage canisters that are used in the storage pools for spent fuel in CLAB will also be disposed of in SFL 4. The planned expansion of storage capacity in CLAB with additional pools is included in this study.

The decommissioning waste from the encapsulation plant is of the same type as the waste from CLAB, but the quantities are much smaller.

The steel cases that will be used for storage of the decommissioning waste are made of 6 mm thick carbon steel and have the outside dimensions 2.4 × 2.4 × 2.4 m, see Figure 2-2. The weight of a steel case is about 1.6 tonnes.

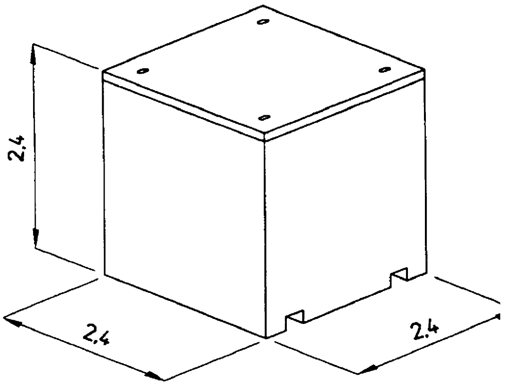

Figure 2-2 Steel case for disposal of decommissioning waste and storage canisters in SFL 4.

2.2.3 SFL 5

Waste with high-induced activity arising during the operation and decommissioning of the NPPs will be disposed of in SFL 5. During the interim period, such waste will be stored in CLAB (about 30–40 years). Small quantities of decommissioning waste from Studsvik will also be disposed of in SLF 5. But the large volumes of LLW that arise in connection with maintenance and decommissioning of the NPPs, and which cannot be decontaminated and released for unrestricted use, are not included in this category. Such waste will be disposed of in SFR 3, which is a planned expansion of the SFR facility at Forsmark.

The waste to SFL 5 will, according to current plans, be packaged in long moulds of reinforced concrete with an inner steel cassette. The waste is placed in the steel cassette, which is placed in the concrete mould and a lid is screwed on. Grouting cement is injected through the lid into the space between the steel cassette and the mould wall, see Figure 2-3. Certain detectors (TIP detectors) with high specific activity must first be placed in lead boxes before they are placed in the steel cassette.

The waste to be sent to SFL 5 consists almost solely of activated metal components with high neutron-induced activity that has arisen due to the fact that they have been installed in or very close to the reactor core, e.g. control rods, moderator tanks, moderator tank covers, core grids and detectors of various kinds. Furthermore, they are surface-contaminated, since they have been in contact with primary reactor coolant.

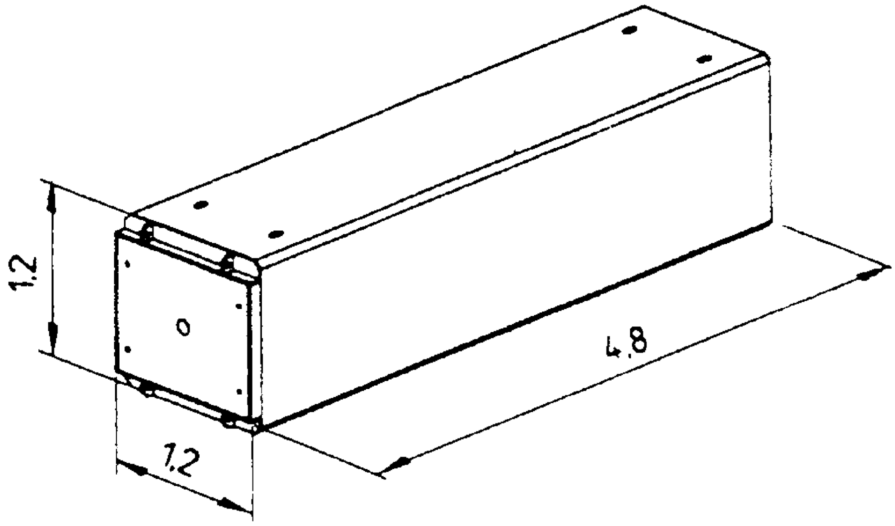

Figure 2-3 Long concrete mould for storage of waste in SFL 5. The walls are 10 cm thick and the weight is 5.5 tonnes.

2.2.4 Summary of waste and packagings in SFL 3-5

According to current forecasts of waste volumes, choice of packagings and conditioning methods, the waste to be sent to SFL 3-5 will occupy a total volume of 25,000 m^3 in the final repository. An approximate breakdown of this waste volume is 20% to SFL 3, 40% to SFL 4 and 40% to SFL 5, see Table 2-1. Waste with different origins and packagings is emplaced in the different repository parts. Figure 2-4 shows the average breakdown of the volume of the packaged waste into waste, packagings, backfill material and void.

The mechanical stability of the waste packages to SFL 4 can be improved by filling the void inside the package with e.g. gravel or concrete. In the safety assessment it is assumed that the void is filled with concrete.

The quantities of materials in the waste and packagings in the different repository parts are given in Table 2-2. The waste consists primarily of inorganic materials such as stainless steel (6,400 tonnes), carbon steel (3,500 tonnes), boron steel (1,300 tonnes) and concrete (3,400 tonnes). Waste containing organic material is emplaced in SFL 3, and current forecasts suggest that there will be about 400 tonnes of organic material. The dominant material in the packagings is concrete, with 12,000 tonnes in containers and 11,000 tonnes as backfill. The quantity of steel in the packagings is 950 tonnes of stainless steel and 2,300 tonnes of carbon steel.

The waste contains some toxic metals such as cadmium (1 tonne), lead (4 tonnes) and beryllium (300 kg). Lead is used e.g. as radiation shielding for some wastes (130 tonnes). Some radionuclides (such as uranium, plutonium and thorium) are moreover chemotoxic, but we assume it is the radiological properties that dominate the toxicity of the waste in these cases (radiotoxicity).

Table 2-1 **Estimated quantities of different main types of waste that will be deposited in SFL 3-5 (operation of NPPs until 2010).**

Origin	Reposi-tory	Packaging	No. of waste packages	Volume in final reposi-tory (m^3)
LILW from Studsvik	SFL 3	drums and moulds	4,500	1,800*
Operational LILW from CLAB and the encapsulation plant	SFL 3	drums and moulds	2,200	3,800
Decommissioning waste from CLAB and the encapsulation plant	SFL 4	steel cases	630	8,700
Transport casks and transport containers	SFL 4	–	80	1,300
Reactor internals and core components	SFL 5	long moulds	1,400	9,700
Decommissioning waste from Studsvik	SFL 5	long moulds	7	50
Total quantity			8,800	25,000

- Forecast based on Studsvik inventory December 1995 (Lindgren *et al.*, 1998)

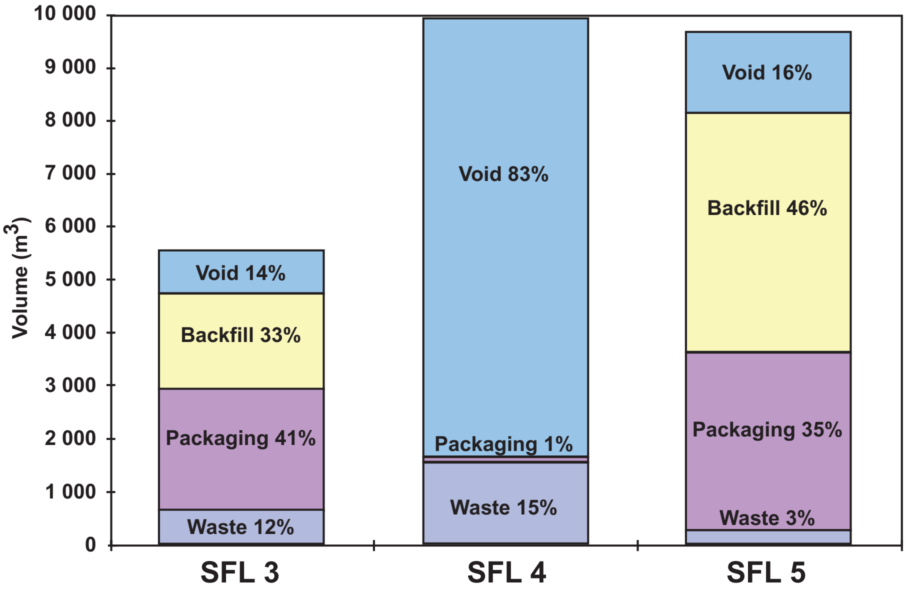

Figure 2-4 *Bar graph showing average breakdown of total volume of waste packages in each repository part into pure waste volume, packaging, backfill material and void.*

Table 2-2 Estimated quantities of different materials (tonnes) in waste and packagings.

	Quantity (tonnes)			
Material	**SFL 3**	**SFL 4**	**SFL 5**	**Total**
Metals				
Aluminium	80			80
Brass and copper	2			2
Cadmium	1			1
Chromium			5	5
Hafnium			4	4
Inconel			2	2
Lead – waste	4			4
Lead – packaging	20		110	130
Stainless steel – waste	70	4,720	1,650	6,440
Stainless steel – packaging	20		930	950
Boron steel – waste		1,260	20	1,280
Carbon steel – waste	80	3,190	280	3,550
Carbon steel – packaging [a)]	820	1,000	440	2,260
Thorium metal	2			2
Zircaloy	3		30	33
Other [b)]	0.1		0.3	0.4
Organic material				
Ion exchange resin	230			230
Paper/wood	5			5
Textiles/rags	20			20
Plastic/rubber	130			130
Unspecified	1			1
Concrete				
Concrete/Cement - waste [c)]	2,580	880		3,460
Concrete – packaging	5,000		7,300	12,300
Concrete – backfill [d)]	700		10,400	11,100
Other [e)]	80			80

[a)] including reinforcement bars in packaging

[b)] zinc and titanium in SFL 3 and beryllium in SFL 5

[c)] stabilized waste and contaminated concrete

[d)] backfill in packagings

[e)] ashes, glass, ferrocyanide precipitates etc.

2.3 Radionuclides in SFL 3-5

2.3.1 SFL 3

The calculated total activity in the waste in SLF 3 in 2040 is $2.2\cdot10^{16}$ Bq, see figure 2-5. The radionuclide content in different waste types has been calculated with the aid of measurement values or correlation factors (Lindgren *et al.*, 1998).

Organic carbon is found in two waste types that contain ion exchange resins; “Decommissioning waste in steel container” and “Ion exchange resins in concrete mould”. The content of organic ^{14}C is estimated to comprise less than 1% of the total quantity of ^{14}C in both of these waste types.

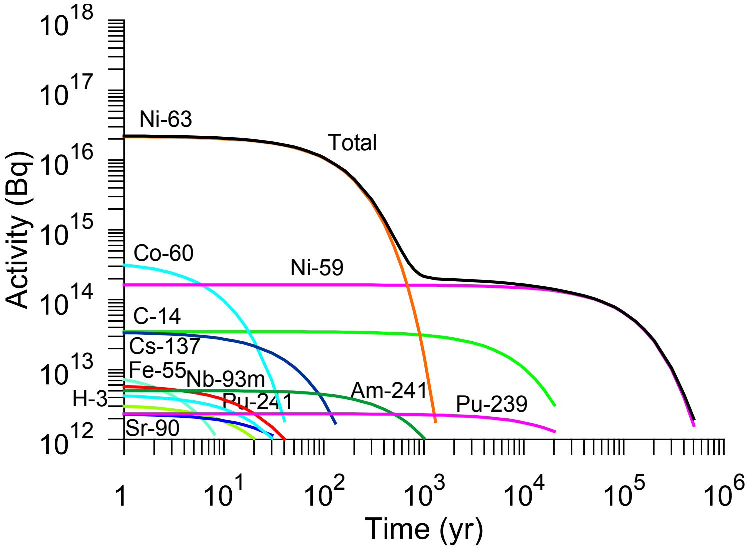

Figure 2-5 Radionuclide activity (Bq) as a function of time in SFL 3. (Time zero is equivalent to the year 2040).

2.3.2 SFL 4

The waste that will be stored in SFL 4 is low-level. The reason it doesn't end up in SFR 1 is that SFR 1 will be closed when CLAB and the encapsulation plant are decommissioned.

The activity in SLF 4 is dominated completely by surface contamination on the storage canisters from CLAB, which have a contaminated surface area of 180,000 m^2. There is relatively little activity on contaminated metal components from the cooling and clean-up systems, since the surface area is limited (about 130 m^2). Surface contamination has been estimated to be 3% of the contamination of reactor components that have been in direct contact with primary reactor coolant. The storage canisters also contain neutron-induced activity, but this activity is much lower than the activity on the surface.

The radionuclide activity in the concrete from the buildings in CLAB and the encapsulation plant derives from limited leakage of pool water and is expected to be relatively low. In this study, the activity in the concrete from CLAB and the encapsulation plant has been equated with estimated activity in structural concrete from Oskarshamn 2.

The activity in the discarded transport casks and containers is judged to be negligible in comparison with the rest of the waste.

The total activity at closure of SFL 4 is estimated to be $7 \cdot 10^{13}$ Bq, of which about 0.1% comes from induced activity, see Figure 2-6 and Figure 2-7. Figure 2-6 has the same scale as the figures showing the activity in SFL 3 and SFL 5, while the scale on the y-axis in Figure 2-7 has been adjusted to show better which radionuclides are present in SFL 4.

The possibility of washing the storage canisters before they are emplaced in SFL 4 and disposing of the washing solution in SFL 3 after treatment and conditioning is being discussed. If all surface contamination on the storage canisters is washed off, the total activity in SFL 4 will decrease to $1 \cdot 10^{11}$ Bq. The total activity after possible decontamination is shown by Figure 2-7.

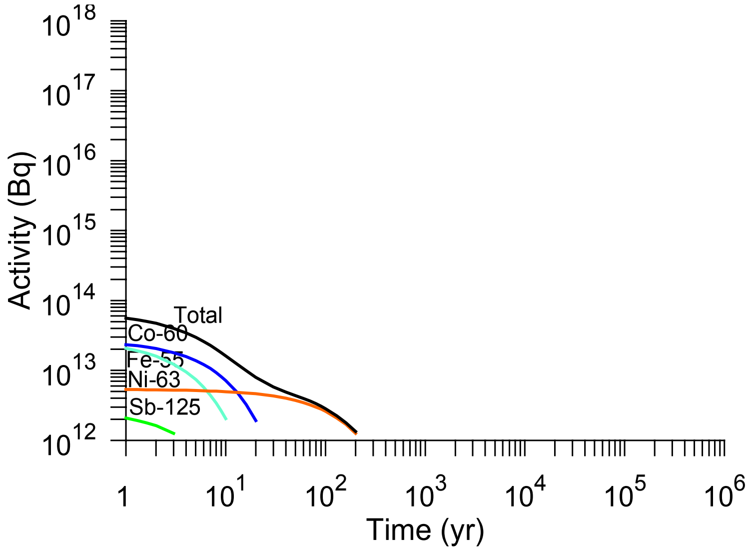

Figure 2-6 Radionuclide activity (Bq) as a function of time in SFL 4. (Time zero is equivalent to the year 2040).

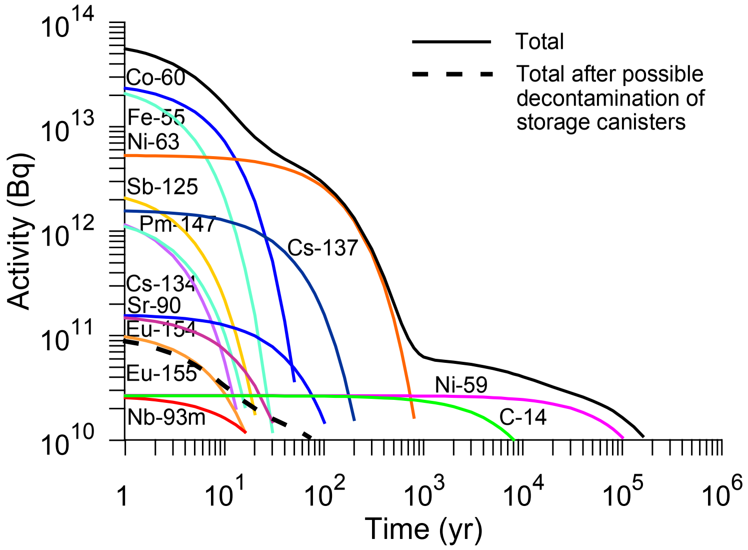

Figure 2-7 Enlargement of radionuclide activity (Bq) as a function of time in SFL 4 and total activity after possible decontamination of storage canisters. (Time zero is equivalent to the year 2040).

2.3.3 SFL 5

The waste in SFL 5 comes from the central portions of the reactor where the materials are exposed to neutron irradiation. The induced activity in the materials declines rapidly with the distance from the core, and surface contamination predominates over the induced activity in the materials a few metres from the core.

The nuclide inventory in the different waste types has been taken into account. For neutron-induced activity, we have used the results of activation calculations wherever possible. Beyond that we have used correlation factors for both neutron-induced activity and surface contamination in order to obtain a complete list of radionuclides in different waste types.

The induced activity dominates the total activity in the waste in SFL 5, which is estimated to be $1.4 \cdot 10^{17}$ Bq in 2040, see Figure 2-8.

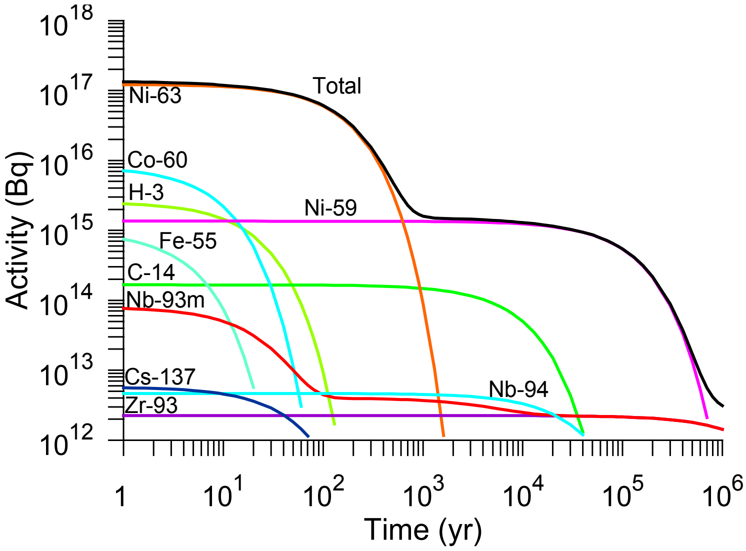

Figure 2-8 Radionuclide activity (Bq) as a function of time in SFL 5. (Time zero is equivalent to the year 2040).

2.3.4 Summary of radionuclide quantities in SFL 3-5

The radionuclide content of SFL 3-5 at closure in 2040 is given in Table 2-3 for the different repository parts. The total activity in SFL 3-5 has been estimated to be $2 \cdot 10^{17}$ Bq. More than 85% of the activity is in SFL 5, 15% in SFL 3, and less than 0.1% in SFL 4. The inventory is dominated by ^{63}Ni, ^{60}Co, ^{3}H and ^{59}Ni. Other nuclides may, however, turn out to be equally important or even more important for the safety of the repository. This is dependent to a high degree on the chemical properties of the nuclide. In this study it has been found that the calculated doses for scenarios with radionuclide release are often dominated by the nuclides ^{36}Cl and ^{93}Mo, see further Chapter 8 and 9. Both of these radionuclides have chemical properties that make them mobile in the barriers and are furthermore so long-lived that they do not decay appreciably during transport from the repository to the biosphere. But relatively speaking, these nuclides constitute only a small portion of the total activity in SFL 3-5.

Table 2-3 Radionuclide inventory (Bq) in SFL 3-5 at closure (2040).

Radionuclide	Half-life (yrs) [a]	SFL 3	SFL 4 [b]	SFL 5	Total
H-3	12	$3.2\cdot10^{12}$	$4.3\cdot10^{9}$	$2.5\cdot10^{15}$	$2.5\cdot10^{15}$
Be-10	$1.6\cdot10^{6}$	$1.2\cdot10^{7}$	$1.6\cdot10^{4}$	$1.4\cdot10^{11}$	$1.4\cdot10^{11}$
C-14	$5.7\cdot10^{3}$	$3.5\cdot10^{13}$	$2.7\cdot10^{10}$	$1.7\cdot10^{14}$	$2.0\cdot10^{14}$
Cl-36	$3.0\cdot10^{5}$	$2.1\cdot10^{10}$	$1.6\cdot10^{7}$	$2.5\cdot10^{11}$	$2.7\cdot10^{11}$
K-40	$1.3\cdot10^{9}$	$1.0\cdot10^{9}$	not calc.	not calc.	$1.0\cdot10^{9}$
Fe-55	2.7	$9.3\cdot10^{12}$	$2.7\cdot10^{13}$	$9.6\cdot10^{14}$	$1.0\cdot10^{15}$
Co-60	5.3	$3.6\cdot10^{14}$	$2.7\cdot10^{13}$	$8.1\cdot10^{15}$	$8.5\cdot10^{15}$
Ni-59	$7.5\cdot10^{4}$	$1.6\cdot10^{14}$	$2.7\cdot10^{10}$	$1.4\cdot10^{15}$	$1.5\cdot10^{15}$
Ni-63	96	$2.2\cdot10^{16}$	$5.3\cdot10^{12}$	$1.2\cdot10^{17}$	$1.4\cdot10^{17}$
Se-79	$6.5\cdot10^{4}$	$4.6\cdot10^{8}$	$6.4\cdot10^{6}$	$4.5\cdot10^{7}$	$5.1\cdot10^{8}$
Sr-90	29	$2.3\cdot10^{12}$	$1.6\cdot10^{11}$	$5.6\cdot10^{11}$	$3.1\cdot10^{12}$
Zr-93	$1.5\cdot10^{6}$	$2.1\cdot10^{10}$	$2.7\cdot10^{7}$	$2.2\cdot10^{12}$	$2.3\cdot10^{12}$
Nb-93m	14	$6.0\cdot10^{12}$	$2.7\cdot10^{10}$	$8.0\cdot10^{13}$	$8.6\cdot10^{13}$
Nb-94	$2.0\cdot10^{4}$	$4.9\cdot10^{11}$	$2.7\cdot10^{8}$	$4.7\cdot10^{12}$	$5.1\cdot10^{12}$
Mo-93	$3.5\cdot10^{3}$	$2.4\cdot10^{11}$	$1.3\cdot10^{8}$	$1.8\cdot10^{12}$	$2.0\cdot10^{12}$
Tc-99	$2.1\cdot10^{5}$	$5.8\cdot10^{11}$	$8.0\cdot10^{9}$	$3.2\cdot10^{11}$	$9.1\cdot10^{11}$
Pd-107	$6.5\cdot10^{6}$	$1.1\cdot10^{8}$	$1.6\cdot10^{6}$	$1.1\cdot10^{7}$	$1.3\cdot10^{8}$
Ag-108m	$1.3\cdot10^{2}$	$1.2\cdot10^{12}$	$1.6\cdot10^{9}$	$9.6\cdot10^{9}$	$1.2\cdot10^{12}$
Cd-113m	14	$5.7\cdot10^{9}$	$9.6\cdot10^{8}$	$1.5\cdot10^{9}$	$8.2\cdot10^{9}$
Sn-126	$1.0\cdot10^{5}$	$5.7\cdot10^{7}$	$8.0\cdot10^{5}$	$5.6\cdot10^{6}$	$6.3\cdot10^{7}$
Sb-125	2.8	$6.6\cdot10^{11}$	$2.7\cdot10^{12}$	$1.4\cdot10^{10}$	$3.3\cdot10^{12}$
I-129	$1.6\cdot10^{7}$	$3.4\cdot10^{7}$	$4.8\cdot10^{5}$	$3.4\cdot10^{6}$	$3.8\cdot10^{7}$
Cs-134	2.1	$3.9\cdot10^{10}$	$1.6\cdot10^{12}$	$7.9\cdot10^{8}$	$1.6\cdot10^{12}$
Cs-135	$2.3\cdot10^{6}$	$5.7\cdot10^{8}$	$8.0\cdot10^{6}$	$5.6\cdot10^{7}$	$6.3\cdot10^{8}$
Cs-137	30	$3.4\cdot10^{13}$	$1.6\cdot10^{12}$	$5.7\cdot10^{12}$	$4.2\cdot10^{13}$
Ba-133	11	$2.9\cdot10^{10}$	$2.7\cdot10^{8}$	$2.9\cdot10^{8}$	$3.0\cdot10^{10}$
Pm-147	2.6	$2.5\cdot10^{10}$	$1.4\cdot10^{12}$	$5.1\cdot10^{9}$	$1.5\cdot10^{12}$
Sm-151	90	$2.3\cdot10^{11}$	$4.8\cdot10^{9}$	$2.7\cdot10^{10}$	$2.6\cdot10^{11}$
Eu-152	13	$1.5\cdot10^{12}$	$1.1\cdot10^{8}$	$1.7\cdot10^{8}$	$1.5\cdot10^{12}$
Eu-154	8.8	$4.8\cdot10^{11}$	$1.6\cdot10^{11}$	$1.1\cdot10^{11}$	$7.6\cdot10^{11}$
Eu-155	5.0	$5.2\cdot10^{10}$	$1.1\cdot10^{11}$	$1.4\cdot10^{10}$	$1.8\cdot10^{11}$
Ho-166m	$1.2\cdot10^{3}$	$8.3\cdot10^{10}$	$1.1\cdot10^{8}$	$7.3\cdot10^{8}$	$8.4\cdot10^{10}$
Pb-210	22	$2.7\cdot10^{11}$	< 1	< 1	$2.7\cdot10^{11}$
Ra-226	$1.6\cdot10^{3}$	$3.8\cdot10^{11}$	< 1	< 1	$3.8\cdot10^{11}$
Ac-227	22	$1.4\cdot10^{6}$	< 1	4.5	$1.4\cdot10^{6}$
Th-229	$7.3\cdot10^{3}$	$1.4\cdot10^{2}$	< 1	< 1	$1.4\cdot10^{2}$
Th-230	$7.7\cdot10^{4}$	$1.8\cdot10^{5}$	2.4	$7.3\cdot10^{1}$	$1.8\cdot10^{5}$
Th-232	$1.4\cdot10^{10}$	$1.1\cdot10^{10}$	< 1	< 1	$1.1\cdot10^{10}$
Pa-231	$3.3\cdot10^{4}$	$2.5\cdot10^{6}$	< 1	7.9	$2.5\cdot10^{6}$
U-232	72	$4.4\cdot10^{6}$	$8.0\cdot10^{2}$	$4.2\cdot10^{3}$	$4.4\cdot10^{6}$
U-233	$1.6\cdot10^{5}$	$3.1\cdot10^{4}$	< 1	$1.3\cdot10^{1}$	$3.1\cdot10^{4}$
U-234	$2.4\cdot10^{5}$	$7.8\cdot10^{8}$	$2.7\cdot10^{4}$	$2.4\cdot10^{5}$	$7.8\cdot10^{8}$
U-235	$7.0\cdot10^{8}$	$6.4\cdot10^{9}$	$5.3\cdot10^{2}$	$3.7\cdot10^{3}$	$6.4\cdot10^{9}$
U-236	$2.3\cdot10^{7}$	$8.1\cdot10^{7}$	$8.0\cdot10^{3}$	$5.6\cdot10^{4}$	$8.1\cdot10^{7}$
U-238	$4.5\cdot10^{9}$	$4.6\cdot10^{10}$	$1.1\cdot10^{4}$	$7.5\cdot10^{4}$	$4.6\cdot10^{10}$
Np-237	$2.1\cdot10^{6}$	$1.8\cdot10^{8}$	$1.1\cdot10^{4}$	$7.9\cdot10^{4}$	$1.8\cdot10^{8}$
Pu-238	88	$3.7\cdot10^{11}$	$1.1\cdot10^{8}$	$5.9\cdot10^{8}$	$3.7\cdot10^{11}$
Pu-239	$2.4\cdot10^{4}$	$2.3\cdot10^{12}$	$8.9\cdot10^{6}$	$6.2\cdot10^{7}$	$2.3\cdot10^{12}$
Pu-240	$6.5\cdot10^{3}$	$1.8\cdot10^{12}$	$1.8\cdot10^{7}$	$1.3\cdot10^{8}$	$1.8\cdot10^{12}$
Pu-241	14	$4.4\cdot10^{12}$	$2.7\cdot10^{9}$	$4.6\cdot10^{9}$	$4.4\cdot10^{12}$
Pu-242	$3.8\cdot10^{5}$	$1.2\cdot10^{9}$	$8.0\cdot10^{4}$	$5.6\cdot10^{5}$	$1.3\cdot10^{9}$
Pu-244	$8.3\cdot10^{7}$	$1.8\cdot10^{2}$	< 1	< 1	$1.8\cdot10^{2}$
Am-241	$4.3\cdot10^{2}$	$5.0\cdot10^{12}$	$2.7\cdot10^{7}$	$6.3\cdot10^{8}$	$5.0\cdot10^{12}$
Am-242m	$1.5\cdot10^{2}$	$2.0\cdot10^{9}$	$2.7\cdot10^{5}$	$1.6\cdot10^{6}$	$2.0\cdot10^{9}$
Am-243	$7.4\cdot10^{3}$	$8.5\cdot10^{9}$	$8.0\cdot10^{5}$	$5.6\cdot10^{6}$	$8.5\cdot10^{9}$
Cm-243	29	$1.7\cdot10^{9}$	$5.3\cdot10^{5}$	$1.8\cdot10^{6}$	$1.7\cdot10^{9}$
Cm-244	18	$4.4\cdot10^{10}$	$8.0\cdot10^{7}$	$1.9\cdot10^{8}$	$4.4\cdot10^{10}$
Cm-245	$8.5\cdot10^{3}$	$7.7\cdot10^{7}$	$8.0\cdot10^{3}$	$5.6\cdot10^{4}$	$7.7\cdot10^{7}$
Cm-246	$4.7\cdot10^{3}$	$2.1\cdot10^{7}$	$2.1\cdot10^{3}$	$1.5\cdot10^{4}$	$2.1\cdot10^{7}$
Total		$2.2\cdot10^{16}$	$6.6\cdot10^{13}$	$1.4\cdot10^{17}$	$1.6\cdot10^{17}$

[a] *ICRP Publication 38. Volumes 11-13. 1983*

[b] *Including surface contamination and induced activity*

2.4 Radiotoxicity of the waste

In order to obtain a rough estimate of the radiotoxicity of the waste to be disposed of in SFL 3-5, the activity (Bq) has been converted to radiotoxicity (Sv) by multiplying the radionuclide activity by the committed effective dose per unit intake (Sv/Bq) according to the formula below. The values for effective dose for the different radionuclides are taken from the IAEA standard (IAEA, 1996, Table II-VI "Ingestion: Committed effective dose per unit intake via ingestion for members of the public").

$$Radiotoxicity = \sum_{i=radionuclide} activity_i \cdot (effective\ \ dose\ per\ unit\ intake)_i$$

Radiotoxicity can be used as a rough measure for comparing radiological risk from different radionuclides, wastes and repository parts. But it is not a measure of the potential load on the biosphere, since no consideration is given to how the radionuclides are released or how they are retarded in the near and far fields.

The total radiotoxicity of the waste in SFL 3-5 is $5 \cdot 10^7$ Sv at closure in 2040. SFL 5 contributes about 86% of this and SFL 3 14%. The contribution from SFL 4 is less than 0.5% and is thereby negligible in comparison with the other repository parts. The dominant radionuclides are ^{63}Ni, ^{60}Co and ^{241}Am in SFL 3, while ^{60}Co and ^{63}Ni dominate in SFL 5. After about 200 years, the radiotoxicity of the waste in SFL 3 will be greater than that in SFL 5 as a consequence of the content of long-lived radionuclides such as ^{241}Am, ^{239}Pu and ^{240}Pu, see Figure 2-9. Ingrowth of ^{210}Pb will cause the radiotoxicity in SFL 3 to increase during the period 10^5–10^6 years.

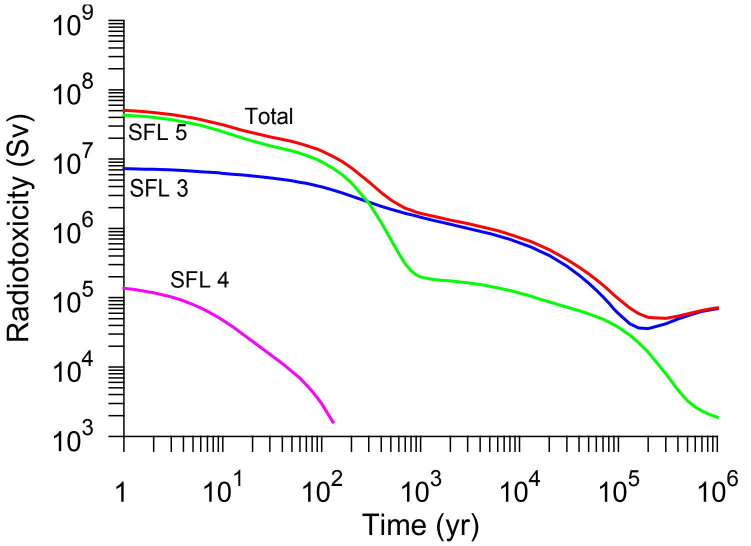

Figure 2-9 Radiotoxicity (Sv) as a function of time in SFL 3-5. (Time zero is equivalent to the year 2040).

2.5 Assumptions and uncertainties

2.5.1 Waste volumes

Most of the waste to be sent to SFL 3 consists of future production of operational waste at CLAB and the encapsulation plant, nearly 70 vol.-%. Studsvik contributes about 30 vol.-% of the waste to SFL 3, and some of this waste already exists today. Information on waste packages from Studsvik produced after 1980 is kept in a waste register, but information on older waste packages is often more scarce. Some older waste already exists but has not yet been packaged. Beyond that, production of waste packages at Studsvik must be estimated from general assumptions regarding future activities and what, for example, the decommissioning of various installations can give rise to.

The decommissioning waste from CLAB and the encapsulation plant, which will be sent to SFL 4, has not yet been produced. The encapsulation plant has not even been built. The quantity of steel and concrete expected to be contaminated during operation of the facilities has therefore been estimated. The nature of the contamination in these facilities at closure will depend above all on the operating conditions, and on the measures adopted to clean contaminated materials. The planned expansion of CLAB from four to eight storage pools has been taken into account in the estimation of the volume of decommissioning waste from CLAB and the number of storage canisters.

The calculated total volume of transport casks and containers in SFL 4 is based on a forecast of how many there will be. However, it is uncertain whether these casks/containers will actually be disposed of in SFL 4. Their contamination level will be very low, and other options than deposition in SFL 4 are possible.

Information that was available regarding a BWR and a PWR has been used to estimate the volume of waste to SFL 5. The quantity of core components and reactor internals from BWRs has probably been somewhat overestimated, since all nine reactors have been equated with the largest reactor type (Oskarshamn 3). Furthermore, we have assumed that the core grid will be replaced in all BWR reactors. The quantity of core components and reactor internals from PWRs, on the other hand, is slightly underestimated, since it is based on the smallest reactor type (Ringhals 2).

2.5.2 Content of radionuclides

Correlation factors have been determined to make it possible to estimate the content of radionuclides in cases where data on quantity are lacking. The lack of such data for a given radionuclide may be due to the fact that that particular radionuclide has been difficult to measure. It may also be due to the fact that for some reason it has never been measured. The method with correlation factors has been used to calculate both surface contamination (54 different radionuclides) and induced activity (16 different radionuclides). To be able to estimate the activity of plutonium (^{239}Pu + ^{240}Pu) and americium (^{241}Am) from the activity of cesium in certain waste types from Studsvik, the same correlation factors have been used as for the ash drums in SFR 1. A review is currently under way which may lead to certain changes.

Both measured and calculated values have been used to arrive at correlation factors. The data come from Swedish facilities, augmented with information from other countries. The correlation factors vary widely, often by several powers of ten, depending on what data they are calculated from. This is not unexpected. In the case of surface contamination, the relationships between different nuclides, and thereby also the correlation factors, are affected by e.g. reactor type, chemistry of the primary coolant, fuel leakage, and impurities in the fuel and corroding materials. In the case of induced activity, the correlation factors are primarily dependent on material composition, occurrence of impurities, irradiation time and neutron fluxes.

The waste register at Studsvik contains data on measurements of radionuclides with a gamma spectrometer. They have been performed in campaigns and cover a limited number of radionuclides. In the case of inhomogeneous waste, the margin of error is estimated at ±70% (Sundgren, 1994). The register covers only waste produced after 1980. The quantity of radionuclides in waste packages produced before 1980 has therefore been estimated by comparison with similar waste produced later. Correlation factors have been used to calculate the content of radionuclides where data are lacking. But the correlation factors have been taken from other types of waste and contaminations, for example from ash drums destined for SFR 1 and from surface contamination in the primary circuits of power reactors. Since the waste from Studsvik has another, sometimes unknown, origin, the calculations are naturally uncertain.

The activity content of the operational waste from CLAB and the encapsulation plant has been estimated from information on the operational waste produced at CLAB. There are uncertainties involved here due to the fact that operating conditions and processes in CLAB may be changed in the future and that experience from encapsulation of fuel is lacking.

The induced activity in the storage canisters from CLAB has been estimated by means of calculations and comparisons with neutron-irradiated reactor waste (correlation factors). The calculations concern storage canisters in which fuel with a higher-than-normal burnup has been stored for 60 years. If the storage time is shorter (e.g. 30 years) and the burnup is normal, the activity will be lower. Another uncertainty stems from the fact that correlation factors have been used which, in turn, derive from other types of waste (reactor waste).

In a similar manner we have calculated the activity on the metal surfaces that have been in contact with pool or other process water in CLAB. The composition of radionuclides has been assumed to be the same as for reactor internals (same correlation factors), but the surface concentration has been assumed to be 3% of that on primary surfaces in the NPPs. The figure is uncertain, but the greatest uncertainty is assumed to lie in the correlation factors used. In SFL 4, the total activity is dominated by the contamination on the storage canisters. It is possible to wash the canisters and transfer the activity to SFL 3 instead, which has better barriers.

Any leakage in the pools or other processes and events during the operation of CLAB could lead to contamination of the structural concrete. We have assumed that the activity will be the same as in the walls and floors in a nuclear power plant, which probably entails an overestimate of the activity in CLAB. The concentration of radionuclides in waste from the encapsulation plant is assumed to be the same as in equivalent waste from CLAB.

Calculations of neutron-induced activity in core components and reactor internals from PWRs have an estimated uncertainty factor of between 5 and 100 (Højerup, 1994). Corresponding data on uncertainty in calculations of induced activity in BWRs are lacking. A Swedish BWR (Oskarshamn 3) and a Swedish PWR (Ringhals 2) have been chosen as models for the other reactors. This increases the uncertainty, since the induced activity in different reactor components is dependent on such parameters as material composition, occurrence of impurities, irradiation time, neutron fluxes and distance to the core.

For core components and reactor internals, the content of induced radionuclides, which has not been included in the activity calculations, has been estimated with the aid of correlation factors to ^{60}Co. The uncertainty is thereby a product of the uncertainty in the correlation factors and the uncertainty in the calculated activity of ^{60}Co.

The contamination on surfaces that have been in contact with the primary reactor coolant has been estimated with a correlation factor and the contamination levels of the key radionuclides (^{60}Co, ^{137}Cs and ^{239}Pu + ^{240}Pu). The uncertainty in the contamination level is judged to be of subordinate importance compared with the uncertainty in the correlation factors.

No allowance has been made for the fact that replaceable components have been taken out of the reactors long before the decommissioning of the reactors. This means that the activities (Bq) of certain relatively short-lived radionuclides – such as ^{55}Fe, ^{60}Co and ^{134}Cs – are overestimated in comparison with more long-lived radionuclides.

In the calculations of the activity content of the waste in SFL 3-5 in 2040, we have used half-lives for the radionuclides as given by the ICRP (1983). Half-lives from Firestone *et al*. (1998) are used in the calculations of radionuclide transport from SFL 3-5, see Chapter 8. For certain radionuclides, different half-lives are given in these two references. This difference in half-life is of no importance for the calculated activity content of long-lived radionuclides. The activity content of short-lived radionuclides is affected, however, although the difference is small and is deemed to be negligible compared with other uncertainties in the calculations of the activity content of the waste.

2.6 References

Firestone R B, Baglin C M, Chu S Y F, 1998. Table of Isotopes, 8th edition, 1998 Update. John Wiley & Sons Inc, New York.

Højerup C F, 1994. Recalculations of neutron induced activations of reactor internals of Ringhals 2. Risø-I-763.

IAEA, 1996. International basic safety standards for protection against ionizing radiation and for the safety of radiation sources. Safety Series No. 115, International Atomic Energy Agency, Vienna.

ICRP, 1983. Annals of the ICRP, Radionuclide transformations, Energy and Intensity of emissions. ICRP Publication 38, Volumes 11-13, Pergamon Press, London.

Lindgren M, Brodén K, Carlsson J, Johansson M, Pers K, 1994. Low and intermediate level waste for SFL 3-5. SKB report AR 94-32, Svensk Kärnbränslehantering AB, Stockholm.

Lindgren M, Pers K, Skagius K, Wiborgh M, Brodén K, Carlsson J, Riggare P, Skogsberg M, 1998. Low and intermediate level waste in SFL 3-5: Reference inventory. A compilation of low and intermediate level waste to be sent to SFL 3-5.
Reg. No 19.41/DL31, Svensk Kärnbränslehantering AB, Stockholm.

Sundgren O, 1994. Gamma-ray counting of medium-activity waste. Radioactivity and radiochemistry, Vol 5, No. 4, page 10-11.

3 Repository design and layout

3.1 Introduction

The design and layout of SFL 3-5 are described in this chapter. This is an updated version that has been made more purpose-suited than the preceding version from 1993 (PLAN 93, 1993). The most important changes are:

- the troughs for moulds in SFL 3 have been replaced with larger disposal pits,
- the three small narrow rock chambers (SFL 5) for core components have been replaced with one wider and longer chamber with the same dimensions and interior design as SFL 3,
- moulds with core components have been emplaced in a concrete enclosure,
- porous concrete is poured around the waste packages in the concrete enclosures in SFL 3 and 5 (previously no grouting at all),
- the rock chambers are backfilled with crushed rock alone instead of a mixture of gravel and bentonite, and
- greater flexibility for altered quantities and types of waste packages has been obtained.

The layout and the functional description are schematic, even though the level of detail may give the opposite impression. The purpose of the detailed information is to provide input data for radionuclide migration calculations.

The premises that govern the layout are described in Section 3.2, and then the layout itself in Section 3.3. Section 3.4 describes the materials in the engineered barriers, i.e. waste containers, concrete structures and backfills of porous concrete and crushed gravel. Section 3.5 deals with conditions in the repository during construction and operation.

3.2 Layout-governing premises

3.2.1 Waste containers

The following compilation shows what types and quantities of waste are to be emplaced in SFL 3-5. Waste and waste packages have been described more fully in Chapter 2. Even if other types of waste packages are used, the design of SFL 3-5 should not be affected more than marginally.

The waste to SFL 3 is packaged in standard concrete moulds (1.2 × 1.2 × 1.2 m) and in 200-litre steel drums, see Section 2.2.1. The maximum weight of a mould, including waste, is 5 tonnes. Approximately 2,800 moulds will be produced, and the average deposition rate is projected to be 150 moulds per year. The drums have a diameter of

0.6 m and a height of 0.9 m. The maximum weight including waste is 500 kg. The total number of drums is projected to be about 4,000, with an average deposition rate of 250 drums/year. Deposition may proceed more rapidly during the first few years, since many of the moulds and drums will already be produced when deposition begins.

The decommissioning waste destined for SFL 4 (including fuel storage canisters from CLAB) is packaged in steel cases (2.4 × 2.4 × 2.4 m), see Section 2.2.2. The maximum weight including waste is about 25 tonnes, and the total number will be about 630. In addition there will be about 80 transport casks and transport containers that cannot be released for unrestricted use. Waste packages to SFL 4 will be produced during an interval of about 2–3 years and deposited as the decommissioning work proceeds. This will occur some time after 2040.

The waste to SFL 5 is cast in long concrete moulds (4.8 × 1.2 × 1.2 m), see Section 2.2.3. The maximum weight is about 25 tonnes and the total number will be over 1,400. These waste packages will be produced during the period 2025 – 2040, entailing an average deposition rate of about 90 moulds per year.

3.2.2 Transportation and transport containers

Ships, road vehicles or rail will be used for transportation to SFL 3-5, depending on where the repository is located. Special vehicles for ramp driving will be used between the ground surface and the repository level. Load frames are required, and the permissible load profile and shipping weights are important parameters.

All waste except the decommissioning waste from CLAB and the encapsulation plant will be transported to the deep repository in special transport containers designed to meet IAEA recommendations. Different types of transport containers will be needed. Two types, type 1 and type 2, have been proposed for SFL 3, see Table 3-1. Type 1 only exists as a conceptual sketch at this point, but type 2, also called ATB-8K, is currently under licensing.

Table 3-1 Different types of transport containers for waste to SFL 3 and SFL 5

Transport container	Type 1 for SFL 3	Type 2 for SFL 3	Type 3 for SFL 5
Capacity	4 moulds or 4 drum trays[b)]	8 moulds or 8 drum trays[b)]	1 long mould
Dimensions[a)]	about 5.5 × 1.9 × 1.7 m	3.25 × 3.05 × 3.0 m	5.5 × 1.9 × 1.7 m
Weight empty	about 45 tonnes	about 74 tonnes	about 45 tonnes
Weight laden	about 65 tonnes	about 114 tonnes	about 70 tonnes
Location of lid	on top	on top	on end

[a)] *length x width x height*
[b)] *4 drums on each drum tray*

LLW to SFL 4 can probably be transported without extra radiation protection. If extra radiation protection is required, suitable transport containers are already available today. Moulds with waste to SFL 5 need a special transport container, however. Type 3 is intended for this purpose, see Table 3-1.

3.3 Layout

3.3.1 General layout

Since the requirements on SFL 3-5 are similar in many ways to those on SFR, the latter facility has served as a model in several respects. This is particularly the case with SFL 3 and SFL 5, where the BMA rock vault in SFR has served as a model. The SFL 3 and SFL 5 rock vaults are identical to each other and parallel. They have been oriented perpendicularly between the entrance tunnel and the building tunnel, see Figure 3-1. Waste is loaded into the tunnels from the entrance tunnel. The building tunnel connects to the repository tunnels at a higher level. The level difference between the entrance tunnel, which is situated on the main level, and the building tunnel is compensated for by the slope of the connecting tunnel.

The transport tunnels that surround the centrally located rock vaults SFL 3 and SFL 5 will be used in the final phase as deposition tunnels for decommissioning waste (SFL 4). After loading is completed, the combined system of transport and deposition tunnels will be separated from the access ramp by means of plugs.

A service building containing power supply switchgear, an electronics room for control of the overhead cranes etc. and a control room is required for operation of SFL 3-5. Personnel quarters for operations personnel and visitors are also needed. The service building has been situated in the forward part of the entrance tunnel.

The entire tunnel system has been given an inclination that permits seeping water to drain towards a low point where pumps are installed to pump the water up to the ground surface.

SFL 3-5 will be built to full capacity in a single stage. Space has, however, been reserved for an additional rock vault if needed.

SFL 3-5 occupies an area of 82,000 m^2 (340 × 240 m). The total rock volume is about 120,000 m^3, excluding access ramp.

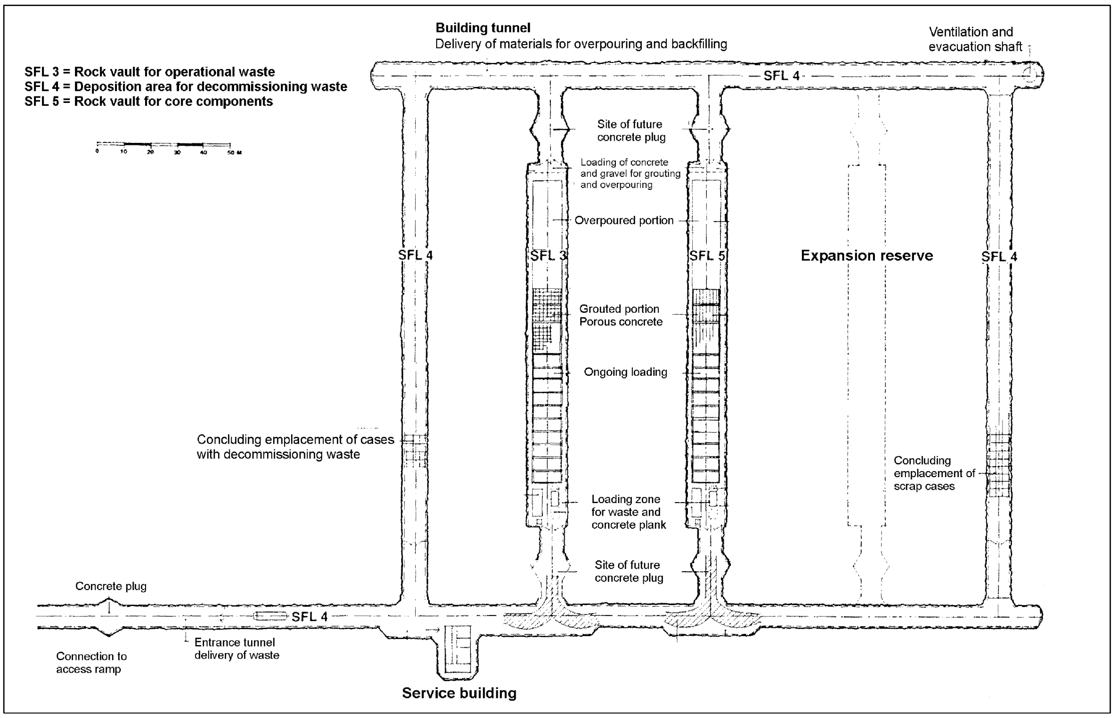

Figure 3-1 General description of rock chambers in SFL 3-5.

3.3.2 SFL 3

The loading section for waste is situated at one end of SFL 3 and a loading section for concrete and backfill material at the other end. The connecting tunnels are long enough to permit casting of concrete plugs after concluded deposition. In this way the rock vault can be isolated from surrounding tunnel systems. The cross-section of the connecting transport tunnels is adapted to the other transport tunnels in the repository, with a load profile that can handle all projected transport needs.

The rock vault is provided with a concrete enclosure that is founded on a compacted bed of stone fill and crushed aggregate, see Figure 3-2. The enclosure is horizontal, while the rock chamber has a slight slope to allow water to drain towards the loading end. The drainage water is evacuated via pipes covered with coarse gravel (>8mm).

The enclosure is divided into three sections, separated by expansion joints that permit movement. Each section is in turn divided by partitions into 7 pits. The purpose of the partitions is to support the longitudinal walls and to help support the emplaced moulds and drums to prevent them from collapsing or moving. This ensures dense packing and effective filling.

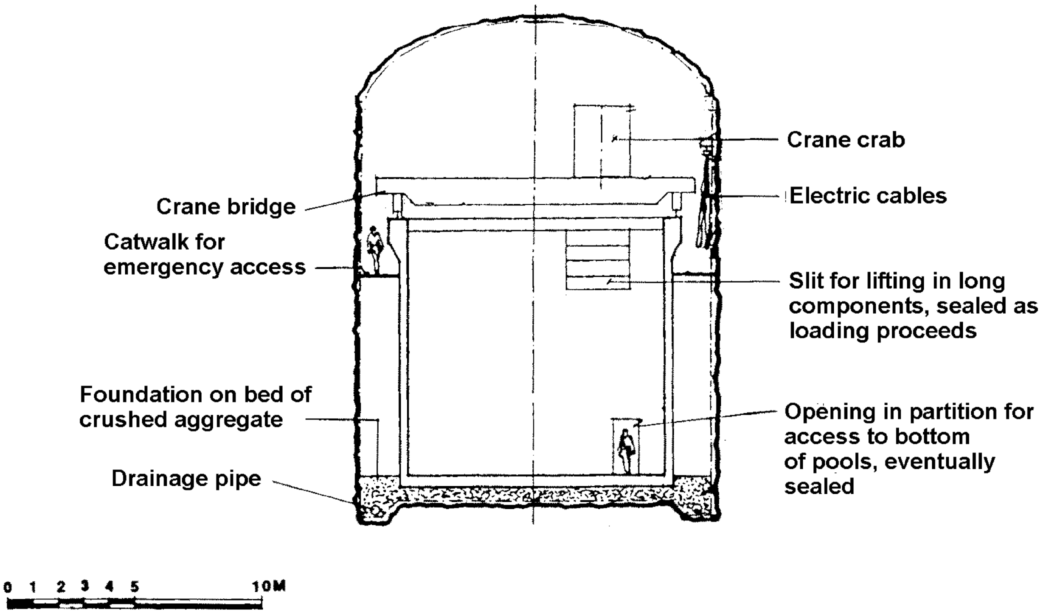

Figure 3-2 SFL 3 and SFL 5, typical section deposition phase.

A maximum of eight moulds or ten drum trays can be stacked on top of each other, see Figure 3-3. Each pit has an inside length of 10.0 m, width 5.0 m and height 10.0 m, and holds 256 moulds or 1,280 drums. It is also possible to place both moulds and drum trays in the same pit, provided that moulds are not placed on top of drum trays. In the outer pits it is possible to place packages with a different shape and size if the partitions are provided with recesses at the top. The waste is loaded in by a remote-controlled crane running on rails resting on the longitudinal outer walls of the enclosure, see Figure 3-2.

The rock vault can be provided with a sheet-metal ceiling to prevent dripping on the waste and water in the sections of the enclosure. Catwalks run along the sides of the enclosure, permitting access to the crane regardless of its position. The catwalks are positioned to permit passage under radiation-shielded conditions.

The waste packages are grouted with porous concrete as deposition progresses. The concrete is mixed in the rear tunnel and pumped through a hose in to the section or carried in a skip by the overhead crane, see Figure 3-4.

When a section is filled, it is covered with a lid consisting of a prefabricated concrete plank, which is brought in via the waste loading section and is lifted into place by means of the crane. A layer of concrete is then poured on top of the concrete plank. Overpouring is done in the same way as grouting, i.e. concrete is mixed and brought in from the rear tunnel connection.

After concluded deposition, a concrete plug is cast in the waste loading section. The rock vault is backfilled with crushed rock which is compacted, after which the other concrete plug is poured so that the rock vault is completely sealed off, see Figure 3-5.

SFL 3's rock vault has a length of about 133 m, a width of 14 m and a height of 19 m. The volume of the rock vault is about 34,000 m^3. The length of the enclosure is about 115 m, the width 11 m and the height 11 m. The volume of concrete in the enclosure is about 3,200 m^3. The number of moulds or drum trays with 4 drums each is 5,376 units.

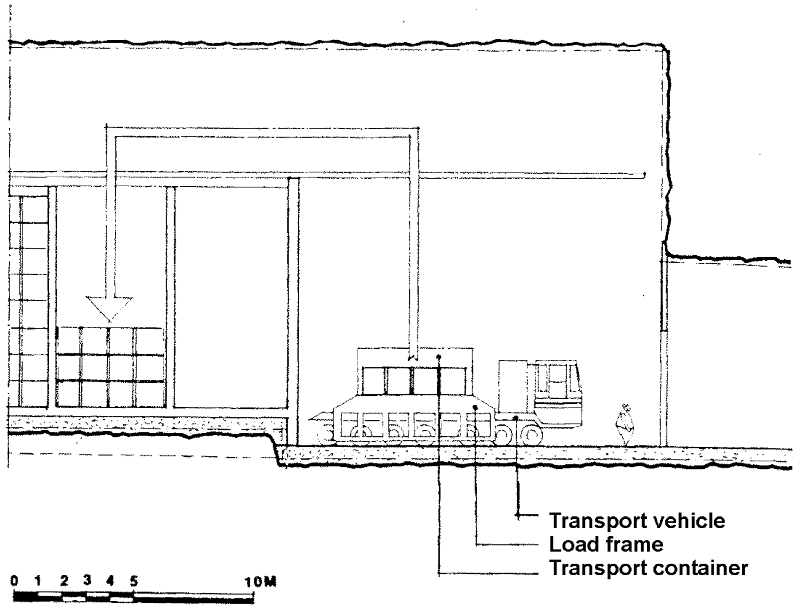

Figure 3-3 SFL 3, loading of moulds, length section.

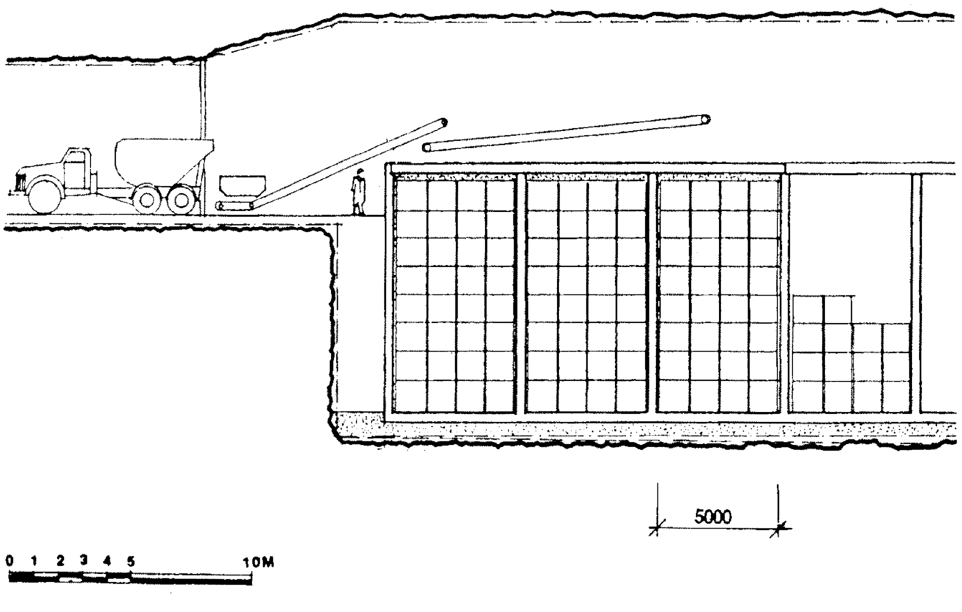

Figure 3-4 SFL 3 and SFL 5, loading of porous concrete for grouting, concrete for covering of pits and of closure material (crushed rock).

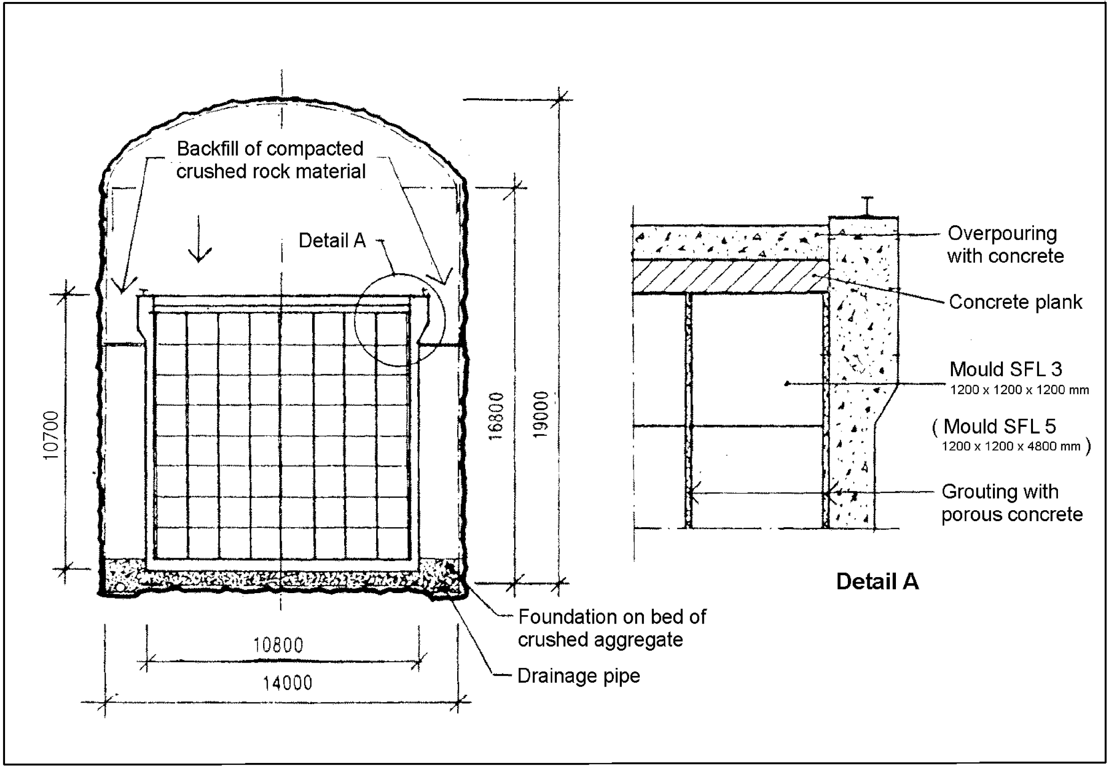

Figure 3-5 SFL 3 and SFL 5, typical section after closure.

3.3.3 SFL 4

The waste containers with decommissioning waste are emplaced in SFL 4, which simply consists of the transport tunnels around SFL 3 and SFL 5. Deposition in SFL 4 can therefore not begin until deposition has been concluded in SFL 3 and SFL 5. The tunnels are separated from other tunnels by concrete plugs.

The tunnels have a width of about 8 m, permitting emplacement of three waste containers abreast, see Figure 3-6. The height of the tunnels is about 6.5 m. About 550 m of tunnel is needed for emplacement of 630 packages. Another 150 m or so of tunnel is needed for other waste. The waste containers are placed directly on the tunnel's roadbed by means of a forklift truck. The road surfacing consists of a reinforced concrete slab on a subcourse of compacted gravel fill. The packages will be packed in crushed rock at loading, whereby the highest possible fill density is striven for.

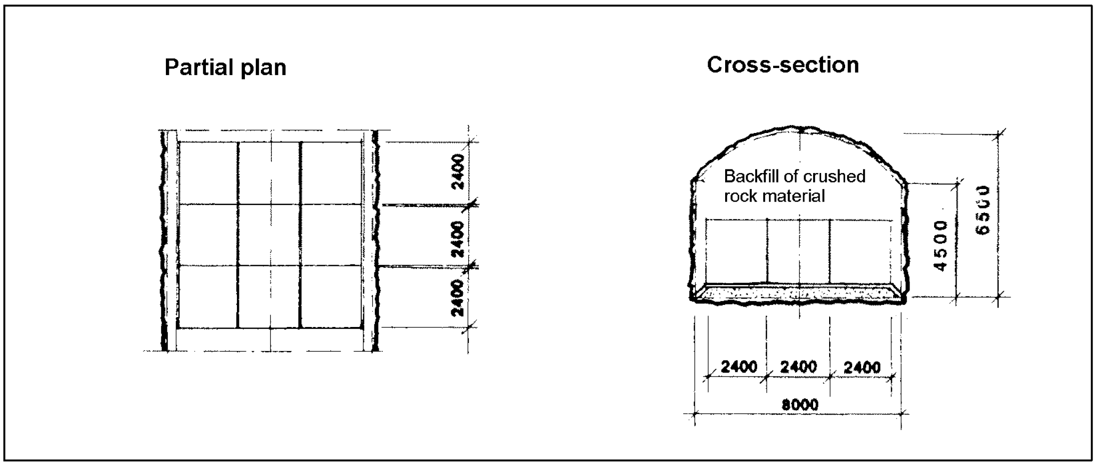

Figure 3-6 SFL 4, plan and typical section after closure.

3.3.4 SFL 5

By choosing a common standard module for operational waste and core components, it is possible to make SFL 5 virtually identical to SFL 3, see Figures 3-2, 3-4, 3-5 and 3-7. SFL 5 holds 64 moulds in each pit, stacked 8 moulds high. There are a total of 21 pits, which means there is room for 1,344 moulds for core components and reactor internals.

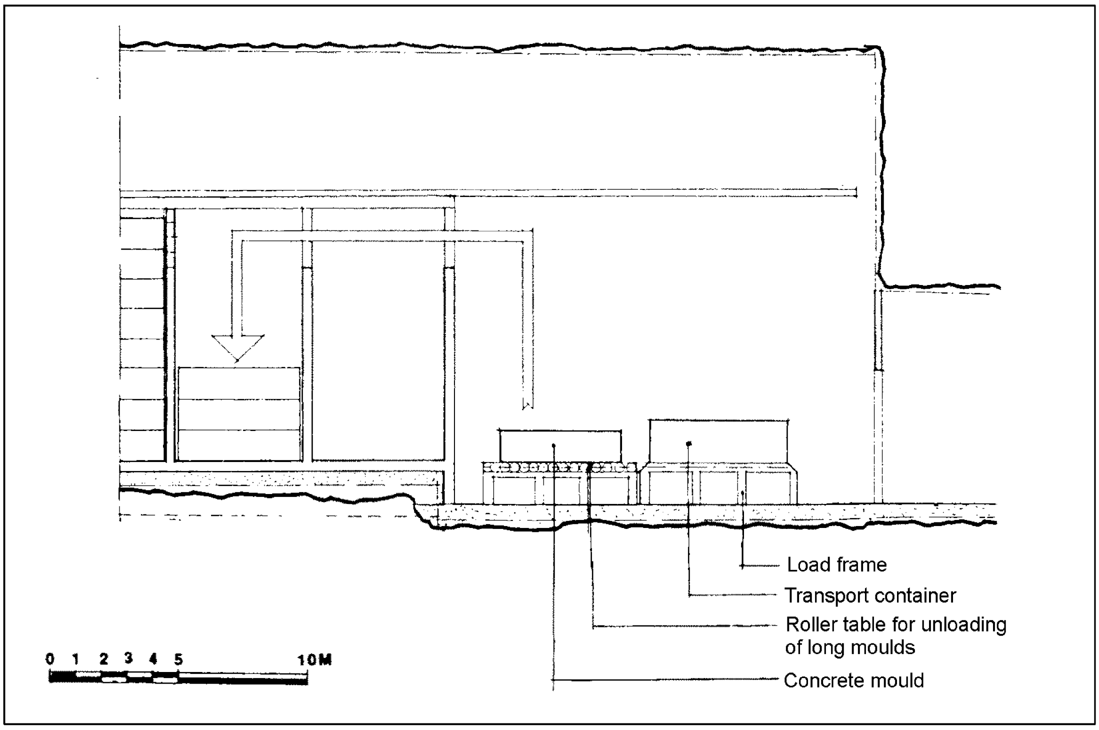

Figure 3-7 SFL 5, loading of long moulds.

3.3.5 Flexibility and provision for expansion

Evacuation routes

Two independent evacuation routes can be used from every place in the repository where people may be present. SFL 3 and SFL 5 have one exit at either end and the transport tunnels, SFL 4, have an exit towards the ramp tunnel and an emergency evacuation route via the ventilation shaft.

Flexibility

The estimated waste volume to SFL 5 is larger today than the available disposal volume in SFL 5, while the waste volume to SFL 3 is smaller than the available disposal volume in SFL 3. Since SFL 3 and SFL 5 are more or less identical, it is possible to redistribute the waste between the rock vaults if needed. If a redistribution between SFL 3 and SFL 5 cannot be permitted for some reason, the lengths of the rock vaults can be adjusted to the actual need when the final requirements have been established. If packages with other contents and other dimensions need to be disposed of, this can be done in the outer pits of SFL 3 and SFL 5, in the same way as in BMA in SFR.

In the sketched layout for SFL 4, approximately 700 m of the available 900 m can be utilized for emplacement of waste. This leaves a margin for accommodating more waste, or the length of the transport tunnels can be reduced. Certain sections of the transport tunnels could be made narrower than 8 m, so that only two cases can be emplaced abreast. In this way any excess tunnel length can be utilized.

Provision for expansion

Space for an additional rock vault has been reserved in the site plan, parallel to the other rock vaults and inside the ring formed by the transport tunnels, see Figure 3-1. The repository can be expanded even more by building yet another rock chamber outside the original ring tunnel. The space in SFL 4 can be increased by lengthening the transport tunnels.

3.4 Barrier properties

The engineered barriers in SFL 3-5 consist of waste containers, concrete structures and backfills of porous concrete and gravel. The physical and chemical properties of the barrier materials and their long-term stability are discussed in this section and in Section 6.7. The influence of the materials on repository conditions is discussed in Chapter 6.

3.4.1 Concrete

The most common cement used in Sweden is Degerhamn Standard Portland Cement (SPC), which is of the type Ordinary Portland Cement (OPC). This cement type is sulphate-resistant due to its low content of aluminium (Lagerblad and Trägårdh, 1994).

There are other types such as slag-containing cement, etc. Almost all cement types are based on Portland cement with different solid additives. When water is added, virtually the same hydration products are obtained in all types, but in different proportions.

We have chosen Degerhamn SPC (which has also been used as a structural material in SFR) as a reference. The cement clinker in SPC consists primarily of tricalcium silicate (51 weight percent), dicalcium silicate (25 weight percent), calcium aluminoferrite (14 weight percent) and calcium aluminate (1.2 weight percent) (Lagerblad and Trägårdh, 1994). When ground cement clinker is mixed with water, cement paste is formed. It stiffens and hardens when the cement particles are hydrated. The hydration products start to grow on the cement particles and finally fill the space between the particles. A residue of unhydrated cement is generally left, even if the amount of water is sufficient to bring about a complete reaction. The most important hydration products in portland cement are calcium silicate hydrates and calcium hydroxide (portlandite). The calcium silicate hydrates form a stiff gel that binds the particles of sand and gravel together in hardened concrete.

Various additives are used in mixing cement and water to improve the properties of the cement, for example sulphonated melamine-formaldehyde polymers, which increase the plasticity of the cement. The total concentration of additives is normally less than 3% of the weight of the cement (Andersson and Allard, 1986).

Concrete is a mixture of cement and sand. The sand, also called aggregate, consists of quartz and a varying number of other silicate minerals. The aggregate comes in different particle sizes, from fine sand to stone. The concrete used in SFR moulds contains 420 kg cement paste/m^3 concrete (Höglund and Bengtsson, 1991), which is equivalent to about 16–18 weight percent.

The pore water in the cement can be analyzed, see Table 3-2. The analyses show that readily soluble alkali hydroxides in fresh cement cause a high pH (about 13.5) in the pore water. If the alkali hydroxides are leached out and disappear, the pH will be determined by the dissolution of portlandite (calcium hydroxide $Ca(OH)_2$), giving a pH of around 12.5. Examples of this are found in the analyses of water used to leach an already leached cement, see Table 3-2.

If the portlandite is also leached out and disappears, the pH will be determined by incongruent dissolution of the cement gel and precipitation of secondary calcium silicate hydrate phases. A gradual lowering of pH is expected due to the change in the ratio between calcium and silicate from 1.7 (pH about 12.5) to 0.85 (pH about 11) (Atkins and Glasser, 1992). This period is followed by congruent dissolution of the remaining gel, which buffers the pH to around 10.

Table 3-2 Analysis of cement pore water from fresh versus leached-out cement. Portland cement and ion concentrations in mmol/l.

	Fresh cement a)	Leached cement b)
Na	28	3 c)
K	83	0.1 c)
Ca	0.9	20
Al	0.04	0.002
Si	0.8	0.003
SO_4	0.04	0.02
Cl	<0.06	2 c)
OH	114	36
pH	13.1	12.6

a) pressed-out pore water (Lagerblad and Trägårdh, 1994)
b) crushed cement, analysis of leachate (Engkvist *et al.*, 1996)
c) same concentration as in original leachate

3.4.2 Gravel

The rock spoils that are hauled away during excavation of tunnels and galleries in SFL 3-5 will be crushed and sieved. A sieve fraction of 4–32 mm (crushed aggregate) will be used as backfill material in the repository. This material has a composition similar to that of the rock, but due to the small size of the particles it has high water permeability, creating a hydraulic cage around the inner parts of the repository. The backfill will also support the inner structures and the tunnel walls.

When it comes to water chemistry, the gravel particles will contribute with their large surface areas to pH- and Eh-buffering reactions, for example consumption of hydroxide from concrete leaching and consumption of oxygen trapped at closure. The gravel will also act as a barrier to the escape of radionuclides. Dissolved radionuclides are sorbed onto the gravel particles or diffuse into and are sorbed in the micropores.

3.5 Repository conditions during construction and operation

The following description focuses on the chemical conditions expected to prevail in the repository during construction and operation. A water saturation phase then ensues immediately upon closure of the repository. See Chapter 6 for a description of the long-range evolution of the repository.

It will take several years to build the repository. This will be followed by operation of the repository, which is expected to last for up to 40 years. Closure will also take a few years. Altogether, the repository will therefore be open for around 50 years. Drainage and ventilation during this time will disturb the natural hydrochemical conditions around the repository. Residual materials and exhaust gases will further affect conditions.

3.5.1 Disturbances in groundwater chemistry

Drainage of the repository during construction and operation will cause an increased downflow of water from the surface and an upflow of groundwater from deeper levels. Surface water contains dissolved oxygen, which is normally consumed by bacteria as the water moves down through the soil layer. The bacteria have the ability to increase their activity if the downflow increases and can thereby consume all or most of the oxygen. Such experiments have been performed in the Äspö HRL (Banwart *et al.*, 1993). The oxygen concentration was measured in the water flowing in from a subvertical fracture zone adjacent to a drained tunnel. The water remained oxygen-free thanks to the fact that the bacteria increased their activity. We therefore did not expect any extensive oxidative weathering of overlying rock that would be of importance for the composition of the groundwater or the barrier function of the rock.

The surface water may be acidified due to air pollution. This does not affect the groundwater at great depths, since the rock has a considerable buffering capacity for pH. This capacity is expected to be sufficient to handle an increased downflow of acid surface water during the period the repository is drained.

Groundwater from great depths has a higher concentration of dissolved salts, compared with groundwater close to the surface. The most widely occurring ions in deep groundwater are Ca^{2+}, Na^{+} and Cl^{-}. A 1,700 m deep hole has been drilled at Laxemar, near the Äspö HRL, and samples have been taken of the groundwater from different levels (Laaksoharju *et al.*, 1995). Relatively high concentrations of chloride, 0.3 mol/l, were encountered at 1,000 m depth and the concentration increased to 1.3 mol/l at 1,700 m. In comparison, seawater has a chloride concentration of 0.5 mol/l.

3.5.2 Influence of air on rock

Air will naturally be admitted to the repository as long as it is open. The rock will, however, be fairly well protected against oxidation by the constant inflow of groundwater containing reducing substances such as Fe^{2+}.

Microbial growth can be expected on wet surfaces such as tunnel walls and roofs (Pedersen and Karlsson, 1995). This will increase the amount of organic material in the repository.

3.5.3 Influence of air on steel and concrete

The influx of air and the moist conditions will cause corrosion of unprotected steel components. The process will be accelerated if saline groundwater penetrates into the repository. Carbon dioxide in the exhaust gases from vehicles and machines as well as in the ventilation air will cause carbonation of exposed concrete surfaces. The reaction consists in the conversion of calcium hydroxide (portlandite) in concrete to calcium carbonate (calcite) by carbon dioxide. The penetration depth of this reaction in concrete is about 5 mm after 50 years (Lagerblad and Trägårdh, 1994).

3.5.4 Expected conditions at closure

No great chemical changes of the waste, the concrete or the backfill material are expected during the 40–50 year period for construction and operation of the repository. Drainage of the repository will prevent groundwater from reaching the concrete and the waste containers. The following processes are foreseen:

- The salinity of the groundwater at repository depth may be altered during the open period, for example increased downflow of surface water may reduce salinity, while increased upflow of groundwater from levels below the repository may increase it.
- The buffering properties of the rock will keep the pH and Eh of the groundwater at repository level roughly unchanged.
- Microbial growth on wet rock walls and roofs will increase the amount of organic material in the repository. The increase will be marginal compared with that already present.
- Unprotected steel components will corrode in the humid air, faster if the groundwater is saline.
- Unprotected concrete surfaces will be carbonated by carbon dioxide from exhaust gases.

3.6 References

Andersson K, Allard B, 1986. The chemical conditions within a cement-containing radioactive waste repository. SKB report SFR 86-08, Svensk Kärnbränslehantering AB, Stockholm.

Atkins M, Glasser F P, 1992. Application of Portland cement-based materials to radioactive waste immobilization. Waste Management, 12, 105-131.

Banwart S, Tullborg E-L, Pedersen K, Gustafsson E, Laaksoharju M, Nilsson A-C, Wallin B, Wikberg P, 1993. Organic carbon oxidation induced by large scale shallow water intrusion into a vertical fracture zone at the Äspö hard rock laboratory. In Migration 93, Charleston SC, USA.

Engkvist I, Albinsson Y, Johansson-Engkvist W, 1996. The long-term stability of cement - Leaching tests. SKB Technical Report TR 96-09, Svensk Kärnbränslehantering AB, Stockholm.

Höglund LO, Bengtsson A, 1991. Some chemical and physical processes related to the long-term performance of the SFR repository. SKB report SFR 91-06, Svensk Kärnbränslehantering AB, Stockholm.

Laaksoharju M, Smellie J, Nilsson A-C, Skårman Ch, 1995. Groundwater sampling and chemical characterisation of the Laxemar deep borehole KLX02. SKB Technical Report TR 95-05, Svensk Kärnbränslehantering AB, Stockholm.

Lagerblad B, Trägårdh J, 1994. Conceptual model for concrete long time degradation in a deep nuclear waste repository. SKB Technical Report TR 95-21, Svensk Kärnbränslehantering AB, Stockholm.

Pedersen K, Karlsson F, 1995. Investigation of subterranean microorganisms. Their importance for performance assessment of radioactive waste disposal. SKB Technical Report TR 95-10, Svensk Kärnbränslehantering AB, Stockholm.

PLAN 93, 1993. Costs for management of the radioactive waste products from nuclear power production. SKB Technical Report 93-28, Svensk Kärnbränslehantering AB, Stockholm.

4 Positioning of SFL 3-5 and properties of repository sites

4.1 Introduction

In the preliminary safety assessment, the repository for long-lived LILW, SFL 3-5, has been located on the same site as the deep repository for spent fuel. In this manner we have been able to choose the same premises as for the more comprehensive and thorough safety assessment of repository for spent fuel and also to utilize some of its results. The safety assessment of the deep repository for spent fuel is called SR 97 (SKB, 1999a) and has been pursued in parallel with the assessment of SFL 3-5. In connection with the future selection of repository sites in Sweden, freedom of choice naturally exists when it comes to the positioning of both the repositories. Other alternatives are also conceivable, such as locating SFL 3-5 at SFR or building it as a completely separate repository. This does not affect the fundamental reasoning and conclusions in this report.

The three hypothetical repository sites included in SR 97 are named Aberg (Äspö), Beberg (Finnsjön) and Ceberg (Gideå), see Figure 4-1. This chapter summarily describes the properties of the repository sites (SKB, 1999b) and how SFL 3-5 has been located together with the deep repository for spent fuel on the different sites (Munier *et al.*, 1997).

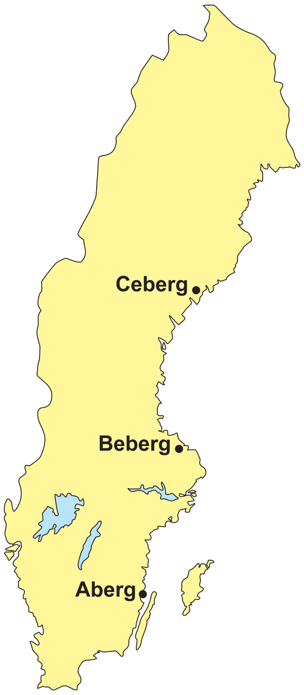

Figure 4-1 The three hypothetical repository sites Aberg (Äspö), Beberg (Finnsjön) and Ceberg (Gideå).

4.2 Positioning of SFL 3-5

The three sites Aberg, Beberg and Ceberg contain rock volumes in which the deep repository for spent fuel and SFL 3-5 can be located (Munier *et al.*, 1997). These rock volumes are bounded by fracture zones. A rock volume that is suitable for a repository must be large enough so that the distance to large fracture zones includes a certain "respect distance". If the rock volumes are small, the repository can be divided up into several volumes, either horizontally or vertically (e.g. disposal on two levels). The SFL 3-5 repository is, however, so small that rock volumes that are acceptable in terms of respect distance and size have been identified on all three sites.

Since the repositories contain different materials, e.g. copper and bentonite in the deep repository for spent fuel compared with concrete and organic matter in SFL 3-5, they are kept separate so that the different materials will not be able to interact. A distance of 1 km has been chosen as the minimum separation distance. Fresh concrete in particular contains readily soluble substances that can increase the pH of the groundwater. To further reduce the likelihood of a groundwater flow from SFL 3-5 to the deep repository for spent fuel, the regional flow direction has therefore been taken into consideration in positioning SFL 3-5.

In Aberg, SFL 3-5 is located at a depth of 300 m below Ävrö. The site is situated about 2 km east of Äspö and 1 km east of the deep repository for spent fuel, see Figure 4-2.

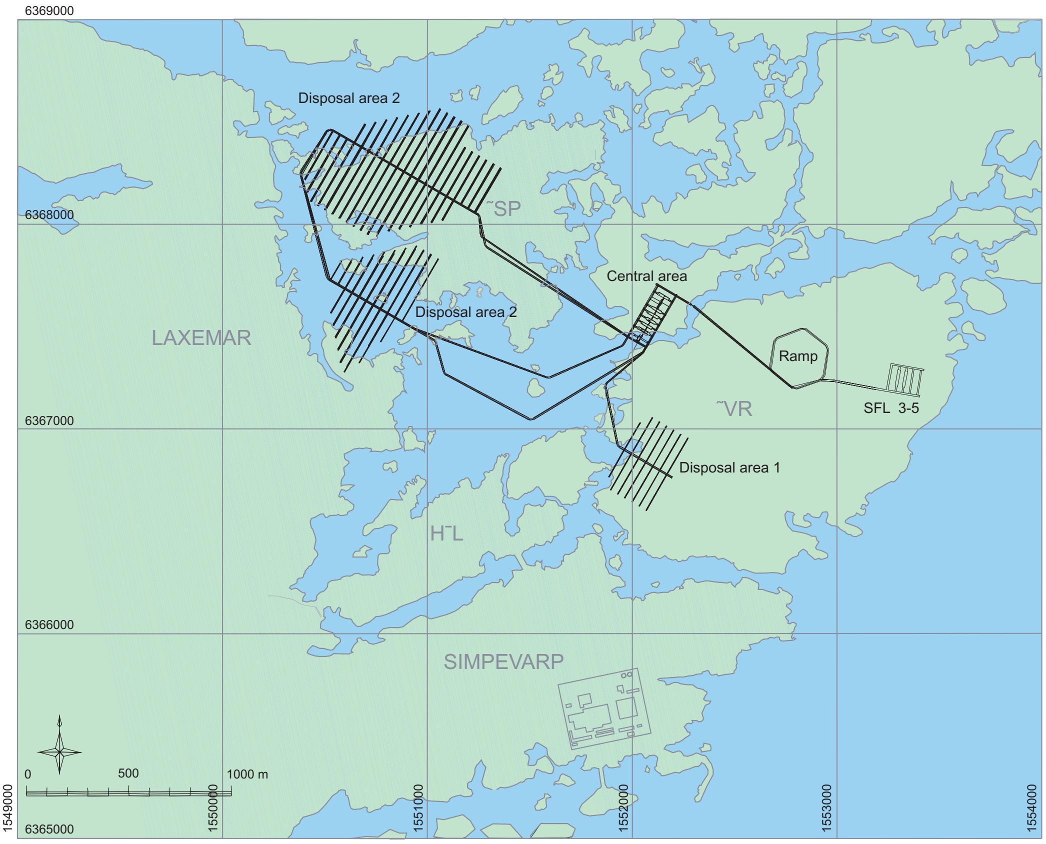

Figure 4-2 Positioning of SFL 3-5 in Aberg.

In Beberg, SFL 3-5 is located at a depth of 360 m, about one kilometre northeast of the deep repository for spent fuel, see Figure 4-3. The repository area is situated immediately north of Finnsjön.

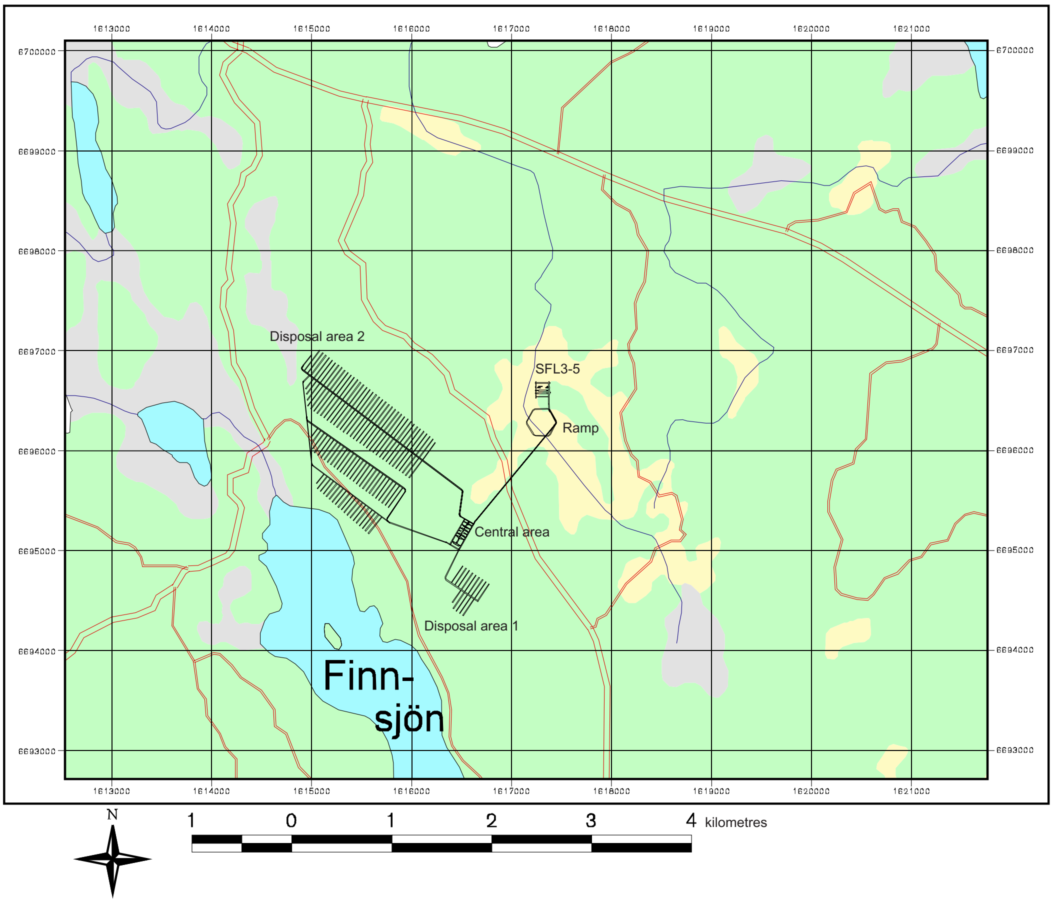

Figure 4-3 Positioning of SFL 3-5 in Beberg.

In Ceberg, SFL 3-5 is located at a depth of 375 m, about one kilometre east of the deep repository for spent fuel. The repository area is situated between the Gideälv and Husån rivers, see Figure 4-4.

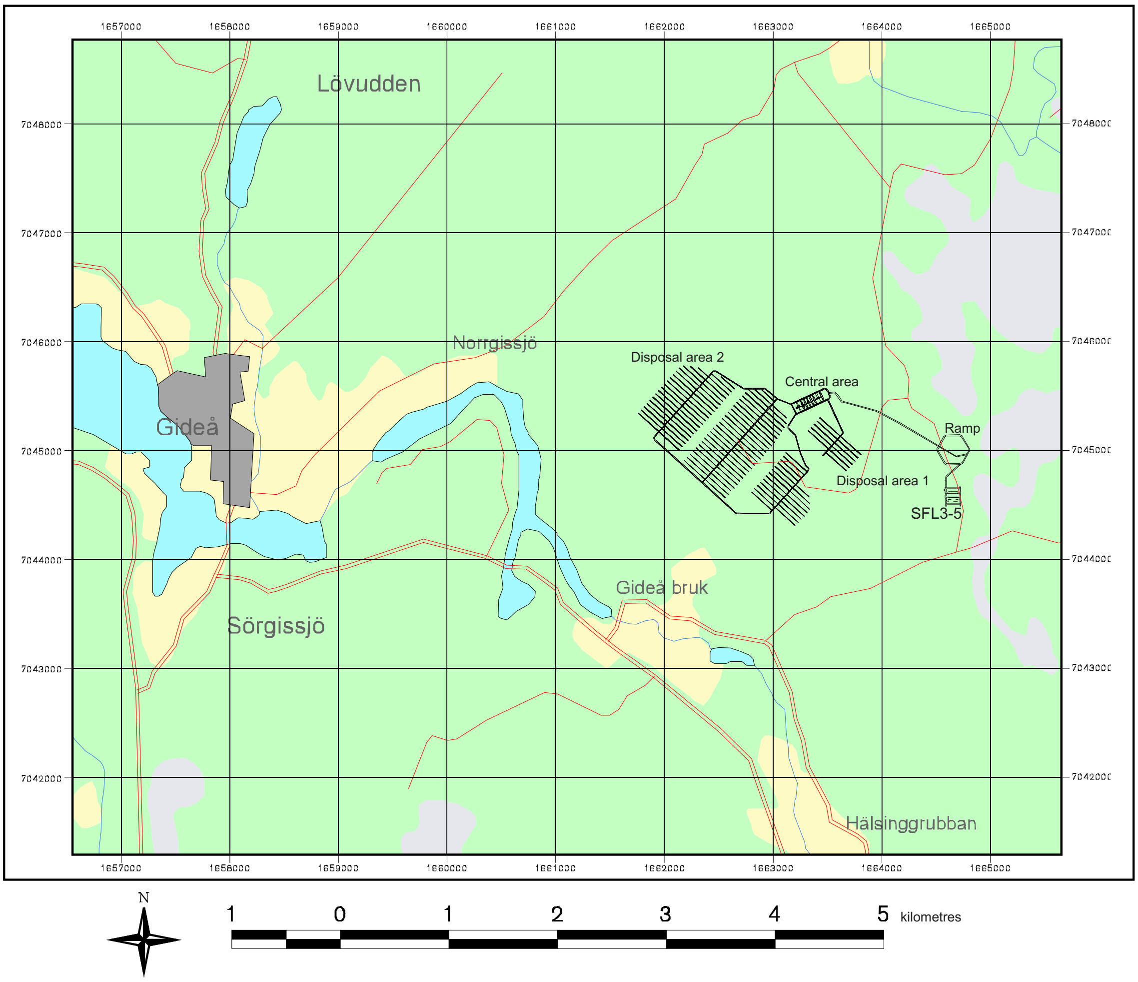

Figure 4-4 Positioning of SFL 3-5 in Ceberg.

4.3 Properties of repository sites

The study areas on the three sites are chosen primarily with the deep repository for spent fuel in mind. They are slightly too small to accommodate SFL 3-5 as well. It has therefore not been possible to utilize the detailed information on the study areas in the assessment of SFL 3-5, where we have instead had to utilize information on the properties of the repository sites on a regional scale. This information is, however, more than adequate to achieve the main purpose of the preliminary safety assessment of SFL 3-5, which is to evaluate the design and the general importance of choice of site. The following text briefly describes the geology, hydrogeology, water composition and temperature on the different sites.

4.3.1 Geology

Aberg

Äspö is an island situated about 40 kilometres north of Oskarshamn, off the Baltic Sea coast of the province of Småland in southern Sweden. The island is located near the mainland in the inner archipelago. The area around Äspö is low-lying, near-coast terrain with an elevation of about 60 m above sea level. Äspö and Ävrö are low-lying, wooded islands with a thin or non-existent soil layer and with bogs or fens in hollows and valleys. The rock types in the Äspö area belong to the approximately 1.65-1.8-billion-year-old transscandinavian granite and porphyry belt (Larsson and Tullborg, 1994).

The region is dominated by a nearly orthogonal north-southerly and east-westerly system of steeply dipping tectonic structures with a horizontal extent of more than 10 km. A local system of northeasterly-trending steeply-dipping structures cuts across Simpevarp and Äspö and is documented in both outcrops and boreholes. The north-southerly structures are presumably younger than the east-westerly oriented ones (Nisca, 1987; Tirén *et al.*, 1987; Tirén and Beckholmen, 1988). The north-southerly structures are considered to be the most water-bearing.

Beberg

The Finnsjö area is situated on the mainland near the northern coast of Uppland County just north of Österbybruk in a relatively low-lying terrain with flat outcrops, mires and small lakes. The average elevation in the Finnsjö region is about 30 metres above sea level. The direction of glacial ice movement in the area varies between north and northeast and has left behind a layer of glacial clay and till a couple of metres thick. The area is covered by about 15% outcrops, 25% mires and 45% moraine, with a decreasing proportion of mire in the southern part (Carlsson and Gidlund, 1983).

The oldest rock types in the area consist of acid volcanites and sedimentary rock with ore-bearing layers (Dannemora Iron Mine). They constitute approximately 25% of the bedrock in the area and occur e.g. along regional shear zones, often trending NW-SE (Ahlbom and Tirén, 1991). More than 50% of the bedrock consists of granodiorite. There are also younger granites in the northern part of the region. The latter rock types are accompanied by pegmatites and aplites.

The lineaments on a regional scale are dominated by two directions of shear zones (Ahlbom and Tirén, 1991) and form a block-like network. One of the regional lineaments, the Singö zone, has been studied in detail in conjunction with the investigations for SFR in Forsmark. The Singö zone is a steeply-dipping, roughly 100–200 m wide zone with a complex inner structure. The inner core of the zone is highly fractured.

Ceberg

The study site in Gideå is situated near Örnsköldsvik in Ångermanland, approximately 600 km north of Stockholm and 8 km from the Baltic Sea coast. In contrast to the regional area around Aberg and Beberg, Ceberg has a significant topographical relief with an elevation varying between about 300 m and sea level. The studied repository area, which lies between the Husån and Gideälven rivers, has a relatively even topography, however. It is forested and the soil is overlain by a thin layer of glacial till, with peat bogs and mires in depressions.

The regional geology is dominated by migmatized sedimentary gneiss. The foliation is generally steeply-dipping, with a gentle dip in the western and southwestern portions. South of Gideå there is a body of gneissic granite, and a younger granite is situated west of the area (Rätan).

The regional lineaments are interpreted as being steeply-dipping with a predominantly WNW to NW strike (Ahlbom *et al.*, 1983; Askling, 1997; Walker *et al.*, 1997). WNW lineaments extend more than 100 m. The NW lineaments are weaker and generally shorter.

4.3.2 Hydrogeology

Aberg

The terrain around Aberg can be described as a mosaic of catchment and recharge areas. The Laxemarån and Gerseboån rivers are the two watercourses that bound the coastal area west of Äspö. The runoff from this area is estimated to be about 190 mm/y (Svensson, 1987). Average precipitation in Kalmar County has been measured at 675 mm/y, and evaporation is estimated at 610 mm/y (Nyberg *et al.*, 1995).

The regional groundwater table is strongly influenced by the topography and variations in the level of the Baltic Sea. The water table reaches its annual maximum level in the spring and its minimum level in August. The groundwater flow follows the topographical level difference from the highlands in the west towards the coast in the east. Brackish water intrudes under the fresh groundwater along the coast (Svensson, 1991).

Several studies show that hydraulic conductivity varies with depth (Ahlbom *et al.*, 1991a, 1991b, 1992a, 1992b, 1992c; Rhén and Gustafsson, 1990; Öberg *et al.*, 1994). In Aberg, the SFL 3-5 repository is positioned at a depth of 300 m. The conductivity of the rock mass at this depth is of the order of 10^{-7} m/s, see Table 4-1.

Table 4-1 Hydraulic conductivity of the rock mass in Aberg on a regional scale. The values apply to measurement sections of 100 m down to a depth of 600 m and sections of 300 m below 600 m (Walker *et al.*, 1997)

Elevation (m.a.s.l.)	Log_{10} K (m/s)	K (m/s)	No. of measurements
0 to -200	-6.89 ± 0.92	$1.3 \cdot 10^{-7}$	264
-200 to -400	-6.70 ± 0.42	$2.0 \cdot 10^{-7}$	30
-400 to -600	-6.59 ± 0.62	$2.6 \cdot 10^{-7}$	9
-600 to -2,000	-7.33 ± 0.51	$4.7 \cdot 10^{-8}$	11
0 to -2,000	-6.89 ± 0.86	$1.3 \cdot 10^{-7}$	314

Beberg

The northern part of the region is drained by the Dalälven, Tämnarån, Forsmarksån and Olandsån rivers. The southern part is drained by Lake Mälaren via the Öresundaån and Fyrisån rivers. Average precipitation is estimated at 670 mm/y. Evaporation has been estimated at 380 mm/y in recharge areas and 540 mm/y in discharge areas (Carlsson and Gidlund, 1983).

The groundwater table in the region is relatively flat with a shallow groundwater flow from SW to NE (Andersson *et al.*, 1991). The regional gradient is estimated to be about 0.2–0.3%. Judging from the topography, the groundwater flows towards the NE, from the upstream lakes of Finnsjön, Lissvass and Åkerbysjön, discharging in the area around Skålsjön. Local groundwater systems may also exist (Boghammar *et al.*, 1993).

The hydraulic conductivity of the rock mass on a regional scale, based on SGU's well data and borehole measurements (measurement sections of 3 m), is given in Table 4-2. In Beberg, SFL 3-5 is positioned at a depth of 360 m. According to Table 4-2, the conductivity of the rock mass at this depth is of the order of 10^{-8} m/s.

Table 4-2 Hydraulic conductivity of the rock mass in Beberg on a regional scale. The conductivity values are upscaled to measurement sections of 100 m (Walker *et al.*, 1997).

Elevation (m.a.s.l.)	Log_{10} K (m/s)	K (m/s)	No. of measurements
Above -100	-6.81 ± 0.21	$1.6 \cdot 10^{-7}$	142
-100 to -200	-7.67 ± 0.15	$2.1 \cdot 10^{-8}$	119
-200 to -400	-8.07 ± 0.10	$8.5 \cdot 10^{-9}$	325
Below -400	-7.80 ± 0.04	$1.6 \cdot 10^{-8}$	265

Ceberg

Average precipitation is 765 mm/y and runoff is estimated to be 345 mm/y (Timje, 1983). The mean annual net recharge of surface water to the groundwater system is 10 mm/y (Walker *et al.*, 1997). The seasonal fluctuation in the groundwater table is 0.2 to 3.9 m (Ahlbom *et al.,* 1983). It is reasonable to assume that the regional groundwater system is driven by the topography and runs from the upland regions in the north and west through the repository area towards the Bothnian Sea.

Table 4-3 shows the conductivity of the rock mass upscaled from 25 m to a 100 m scale. In Ceberg, SFL 3-5 is positioned at a depth of 375 m. According to Table 4-3, the conductivity of the rock mass at this depth is of the order of 10^{-10} m/s.

Table 4-3 Hydraulic conductivity of the rock mass in Ceberg on a regional scale. The conductivity values are upscaled to measurement sections of 100 m (Walker *et al.*, 1997).

Elevation (m.a.s.l.)	Log_{10} K (m/s)	K (m/s)
+110 to 0	-7.16 ± 0.38	$6.9 \cdot 10^{-8}$
0 to -100	-8.65 ± 0.70	$2.2 \cdot 10^{-9}$
-100 to -300	-9.63 ± 0.71	$2.3 \cdot 10^{-10}$
Below -300	-9.85 ± 0.99	$1.4 \cdot 10^{-10}$

4.3.3 Water composition

The chemical composition of the deep groundwater on the three sites is represented by samples taken from four boreholes: KAS02 for Aberg, KFI7 and BFI01 for Beberg and KGI04 for Ceberg. This is the same assumption that has been chosen for the safety assessment SR 97, see Table 4-4 (Laaksoharju *et al.*, 1998; Andersson, 1999). The water samples are taken at depths of between 300 and 600 m, which matches the levels required for deep disposal.

Aberg's water from the KAS02 borehole is saline, lime-poor and sulphate-rich, see Table 4-4.

Beberg has two water compositions: a saline, lime-poor and sulphate-rich water (BFI01), and a lime-rich, chloride- and sulphate-poor fresh water (KFI07), see Table 4-4. Despite the fact that the compositions are different, they occur close to each other at the depth in question. The boundary between them runs through a flat hydraulically conductive zone in the rock. It is difficult to determine which composition will dominate in the repository. Drainage pumping during the open period may, for example, be an important factor that will influence the final composition. We therefore include both compositions in the safety assessment of SFL 3-5 in Beberg.

Ceberg's water from the KGI04 borehole is a lime- and sulphate-poor fresh water, see Table 4-4.

Table 4-4 Chemical composition of reference waters. Waters from borehole sections in Äspö (KAS02), Finnsjön (BFI01 and KFI07) and Gideå (KGI04) have been chosen to represent deep groundwater in Aberg, Beberg and Ceberg (Laaksoharju *et al.*, 1998).

Groundwater components	Unit	Aberg	Beberg 1	Beberg 2	Ceberg
Borehole		KAS02	BFI01	KFI07	KGI04
Date of measurement		88-05-04	86-10-27	80-11-19	82-07-04
Depth	m	528	436	508	384
Drilling water content	%	0.19	0.02	–	11.03
Na^+	mg/l	2,100	1,700	275	105
K^+	mg/l	8	13	2	2
Ca^{2+}	mg/l	1,890	1,650	142	21
Mg^{2+}	mg/l	42	110	17	1
Sr^{2+}	mg/l	35	21	–	–
Fe^{2+}	mg/l	0.24	–	1.80	0.05
Mn^{2+}	mg/l	0.29	0.82	0.13	0.01
HCO_3^-	mg/l	10	47	278	18
SO_4^{2-}	mg/l	560	370	49	0.1
Cl^-	mg/l	6,410	5,500	555	178
I^-	mg/l	–	0.12	–	0.14
Br^-	mg/l	40	32	–	–
F^-	mg/l	1.5	1.2	1.5	3.2
HS^-	mg/l	0.15	<0.01 1)	–	<0.01 1)
NH_4^+ calculated as N	mg/l	0.03	0.35	0.09	0.01
NO_3^- calculated as N	mg/l	<0.010 1)	<0.005 1)	<0.002 1)	0.009
NO_2^- calculated as N	mg/l	<0.001 1)	0.005	0.010	<0.001 1)
$PO_{4.tot}$ calculated as P	mg/l	0.005	0.005	0.040	0.008
SiO_2 calculated as Si	mg/l	4.1	5.4	5.6	4.7
DOC (dissolved organic carbon)	mg/l	1	–	6	2
Eh 2)	mV	-308	–	-250	-202
pH 2)		7.7	7.0	7.9	9.3

1) below the detection limit
2) see Laaksoharju et al. 1998

4.3.4 Thermal properties

The temperature in Swedish groundwater varies with the latitude. The temperature at a depth of 500 m is estimated to be 7.5–10°C in northern Sweden and 10–15°C in central and southern Sweden. The groundwater temperature in the southernmost parts of the country (Skåne and Blekinge) and on Gotland is estimated to be about 15–20°C (Sundberg, 1995).

The average temperature gradient in Aberg is estimated to be 15.0ºC/km, and a mean temperature of 14.6ºC has been measured at a depth of 500 m (Ahlbom *et al.*, 1995). Based on these values, the mean temperature at the 300 m level, where SFL 3-5 has been placed, is about 11.6ºC.

The average temperature gradient in Beberg is estimated to be 12.7ºC/km, and measurements in boreholes gave a mean temperature of 11.6ºC at a depth of 500 m in the rock mass (Ahlbom *et al.*, 1995). In Beberg, SFL 3-5 is positioned at a depth of 360 m, entailing a mean temperature of about 10.9ºC.

The mean temperature at a depth of 500 m in Ceberg is 10.9ºC, and the temperature gradient is 15.5ºC/km (Ahlbom *et al.*, 1995). Based on these values, the mean temperature at a level of 375 m, where SFL 3-5 has been placed, is about 9ºC.

Thermal conductivity is governed by the mineral composition of the rock type, particularly its quartz content. In acid rocks with a granitic composition, for example, it is calculated that the thermal conductivity lies between 2.5 and 5.5 W/mºC (Sundberg, 1995). We have used the value 3.5 W/mºC to calculate how much the temperature in SFL 3-5 rises due to radioactive decay, see Section 6.3.

4.4 References

Ahlbom K, Albino B, Carlsson L, Nilsson G, Olsson O, Stenberg L, Timje H, 1983. Evaluation of the geological, geophysical and hydrogeological conditions at Gideå. SKBF/KBS Technical Report TR 83-53, Svensk Kärnbränslehantering AB, Stockholm.

Ahlbom K, Tirén S, 1991. Overviev of geological and geohydrologic conditions at the Finnsjön site and its surroundings. SKB Technical Report TR 91-08, Svensk Kärnbränslehantering AB, Stockholm.

Ahlbom K, Andersson J-E, Nordqvist R, Ljunggren C, Tirén S, Voss C, 1991a. Gideå study site. Scope of activities and main results. SKB Technical Report TR 91-51, Svensk Kärnbränslehantering AB, Stockholm.

Ahlbom K, Andersson J-E, Nordqvist R, Ljunggren C, Tirén S, Voss C, 1991b. Fjällveden study site. Scope of activities and main results. SKB Technical Report TR 91-52, Svensk Kärnbränslehantering AB, Stockholm.

Ahlbom K, Andersson J-E, Nordqvist R, Ittner T, Ljunggren C, Tirén S, 1992a. Klipperås study site. Scope of activities and main results. SKB Technical Report TR 92-22, Svensk Kärnbränslehantering AB, Stockholm.

Ahlbom K, Andersson J-E, Ittner T, Ljunggren C, Tirén S, 1992b. Kamlunge study site. Scope of activities and main results. SKB Technical Report TR 92-15, Svensk Kärnbränslehantering AB, Stockholm.

Ahlbom K, Andersson J-E, Ittner T, Ljunggren C, Tirén S, 1992c. Finnsjön study site. Scope of activities and main results. SKB Technical Report TR 92-33, Svensk Kärnbränslehantering AB, Stockholm.

Ahlbom K, Olsson O, Sehlstedt S, 1995. Temperature conditions in the SKB study sites. SKB Technical Report TR 95-16, Svensk Kärnbränslehantering AB, Stockholm.

Andersson J-E, Nordqvist R, Nyberg G, Smellie J, Tirén S, 1991. Hydrogeological conditions in the Finnsjön area. Compilation of data and conceptual model. SKB Technical Report TR 91-24, Svensk Kärnbränslehantering AB, Stockholm.

Andersson J, 1999, SR 97 – Data and data uncertainties. Compilation of data and evaluation of data uncertainties for radionuclide transport calculations. SKB Technical Report TR 99-09, Svensk Kärnbränslehantering AB, Stockholm.

Askling P, 1997. Gideå lineament map. An interpretation based on elevation data models. SKB Progress Report PR U-97-06, Svensk Kärnbränslehantering AB, Stockholm.

Boghammar A, Grundfelt B, Widén H, 1993. Analysis of the regional groundwater flow in the Finnsjön area. SKB Technical Report TR 93-15, Svensk Kärnbränslehantering AB, Stockholm.

Carlsson L, Gidlund G, 1983. Evaluation of the hydrogeological conditions at Finnsjön. SKBF KBS Technical Report TR 83-56, Svensk Kärnbränslehantering AB, Stockholm.

Laaksoharju M, Gurban I, Skårman, Ch, 1998. Summary of hydrochemical conditions at Aberg, Beberg and Ceberg. SKB Technical Report TR 98-03, Svensk Kärnbränslehantering AB, Stockholm.

Larsson S Å, Tullborg E-L, 1994. Tectonic regimes in the Baltic shield during the last 1200 Ma - a review. SKB Technical Report TR 94-05, Svensk Kärnbränslehantering AB, Stockholm.

Munier R, Sandstedt H, Niland L, 1997. Förslag till principiella utformningar av förvar enligt KBS-3 för Aberg, Beberg och Ceberg. SKB Report R-97-09, Svensk Kärnbränslehantering AB, Stockholm.

Nisca D, 1987. Aerogeophysical interpretation. Bedrock and tectonic analysis. SKB HRL Progress Report PR 25-87-04, Svensk Kärnbränslehantering AB, Stockholm.

Nyberg G, Jönsson S, Ekman L, 1995. Groundwater level program. Report for 1994. SKB HRL Progress Report PR 25-95-08, Svensk Kärnbränslehantering AB, Stockholm.

Rhén I, Gustafsson G, 1990. DDP evaluation of hydrogeological data, Report U(G) 1990/59, Vattenfall, Stockholm.

SKB, 1999a. SR 97 – Main Report, Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1999b. SR 97 – Waste, repository design and sites. Svensk Kärnbränslehantering AB, Stockholm.

Sundberg J, 1995. Termiska egenskaper för kristallint berg i Sverige. Kartor över värmekonduktivitet, värmeflöde, och temperatur på 500 m djup. Projekt rapport PR D-95-018, Svensk Kärnbränslehantering AB, Stockholm.

Svensson T, 1987. Hydrogeological conditions in the Simpevarp area. SKB HRL Progress Report 25-87-09, Svensk Kärnbränslehantering AB, Stockholm.

Svensson U, 1991. Groundwater flow at Äspö and changes due to the excavation of the laboratory. SKB HRL Progress Report PR 25-91-03, Svensk Kärnbränslehantering AB, Stockholm.

Timje H, 1983. Hydrogeologiska undersökningar inom typområdet Gideå. SKBF/KBS rapport AR 83-26, Svensk Kärnbränslehantering AB, Stockholm.

Tirén S, Beckholmen M, Isaksson H, 1987. Structural analysis of digital terrain models, Simpevarp area, Southeastern Sweden. Method study EBBA II. SKB HRL Progress Report PR 25-87-21, Svensk Kärnbränslehantering AB, Stockholm.

Tirén S, Beckholmen M, 1988. Structural analysis of the Simpevarp sea area, Southeastern Sweden. Lineaments and rock blocks. SKB HRL Progress Report 25-88-01, Svensk Kärnbränslehantering AB, Stockholm.

Walker D, Rhén I, Gurban I, 1997. Summary of hydrogeologic conditions at Aberg, Beberg and Ceberg. SKB Technical Report TR 97-23, Svensk Kärnbränslehantering AB, Stockholm.

Öberg A, Saksa P, Ahokas H, Routsalainen P, Snellman M, 1994. Summary report of the experiences from TVO´s site investigations. SKB Technical Report TR 94-17, Svensk Kärnbränslehantering AB, Stockholm.

5 Biosphere

5.1 Introduction

The radiation doses to which the calculated releases of radionuclides to the biosphere from SFL 3-5 can give rise have been calculated in the same way as in the safety assessment of the deep repository for spent fuel (SKB, 1999). It has, however, been necessary to expand the list of radionuclides, since waste in SFL 3-5 contains some radionuclides that do not have to be included in the list for spent fuel, such as ^{93}Mo.

This chapter describes the inflow of radionuclides to the biosphere, models and exposure pathways for the three sites Aberg, Beberg and Ceberg (Bergström *et al.*, 1999 and Nordlinder *et al.*, 1999). The dominant ecosystems, population, meteorology, topography etc. on the three sites are described in another report (Lindborg and Schüldt, 1998).

5.2 Calculation of radionuclide migration and radiation doses

The hydrological calculations show where radionuclides carried by the groundwater can reach the biosphere, see Chapter 7. Typical discharge areas for groundwater are lakes, watercourses, coastal areas, wetlands, agricultural land and forestland. Different typical ecosystems (modules) are chosen for the dominant ecosystems at the discharge points. Two calculation models belong to each typical ecosystem. The first is a transport model for calculating the flux of radionuclides between the different compartments in the module. The other model is used to calculate how much man is exposed to the radiation from the radionuclides.

The transport model gives the content of radionuclides in the various compartments at different points in time (years). The compartments may be e.g. water, soil or sediment. It is assumed that the radionuclides are homogeneously distributed within the compartment. The flow of radionuclides between the compartments is described by transfer coefficients expressed as flux per annum. The transfer of nuclides takes place with water or solid material as the carrier. The transfer coefficients can be calculated with the aid of data from the site (e.g. water turnover) and the properties of the substances (e.g. sorption on minerogenic material). In the calculation, the system receives a continuous input of 1 Bq/y of each radionuclide, and the course of events during 10,000 years is simulated. After that time the system has usually reached a steady state where the continuous inflow is balanced by radioactive decay, losses to sinks such as sediments or export over the system boundary with e.g. a water flow. The situation that has then been established with specific concentrations of radionuclides in the different compartments is the result of the calculation with the transport model. The software package BIOPATH is used for the calculations (Bergström *et al.*, 1982).

The radiation dose to man is calculated from the total quantity of radionuclides inhaled or ingested in the form of contaminated food during one year (internal exposure). The concentrations of the radionuclides in the foodstuffs (e.g. fish, potatoes) are calculated with equilibrium factors that describe the concentration in organisms in relation to the concentration in the compartments where they occur, e.g. soil, sediment or water. The equilibrium factors are dependent on the substances, and in many cases also the organisms, involved. For many nuclides, data are available from field measurements or laboratory studies that can be used to calculate the equilibrium factors, but for some substances we have to derive the equilibrium factors from comparisons with similar substances.

Table 5-1 Consumption of different foodstuffs.

Foodstuff	Intake	Unit
Water	600	l/y
Fish	30	kg/y
Milk	220	l/y
Meat	75	kg/y
Cereals (bread)	70	kg/y
Root crops	110	kg/y
Leafy vegetables	55	kg/y
Crustaceans	2	kg/y
Algae	2	kg/y
Soil	0.01	kg/y

An ecosystem-specific dose conversion factor, EDF, is calculated for each typical ecosystem and each nuclide. It indicates the relationship between activity input to the typical ecosystem in Bq and dose to man in Sv after all exposure pathways have been taken into account. We have thus assumed that people live in one typical ecosystem and eat solely the foodstuffs produced there, see Table 5-1. This means that these people are exposed to the highest dose, since their intake of contaminated foodstuffs is not diluted with uncontaminated foodstuffs. Nor has any allowance been made for the fact that radionuclides are removed by cleaning or food preparation procedures. The dose conversion factors (ICRP) used to calculate dose to man from the activity to which man is exposed are given in Bergström *et al.* (1999).

We have developed calculation models for the typical ecosystems: Lake, Running Water, Coastal Area (open coast and archipelago), Agricultural Land and Peatland. In addition, a model has been developed for exposure from contaminated groundwater via wells. Turnover of radionuclides in forestland cannot been calculated, since data are lacking for all nuclides except cesium. For forestland we have instead used EDFs for peat. The EDFs used to calculate dose to man from the release of radionuclides from SFL 3-5 to the biosphere in Aberg, Beberg and Ceberg are given in Table 8-16 in Section 8.6 and in Table 9-2 in Section 9.4.

5.3 Typical ecosystems

5.3.1 Lake

The lake ecosystem deals with water and sediment. Radionuclides are brought directly into the lake via groundwater or contaminated watercourses. Important factors are sorption of radionuclides on particles in the water, the water turnover rate in the lake, sedimentation and resuspension, uptake in biota (plants and animals) and the productivity of the lake (trophic level).

Man is exposed via consumption of water, fish, crustaceans, and milk and meat from cattle that have grazed at the shore or on areas irrigated with contaminated water. Man and cattle are also assumed to eat food from crops irrigated with contaminated water, such as leafy vegetables, root crops, cereals, concentrated feed and hay. Inhalation of dust and external exposure from the contaminated soil are also included.

5.3.2 Running water

In the typical ecosystem for running water, the radionuclides are assumed to be homogeneously distributed in the water and sedimentation is not taken into account. Important parameters in the transport calculations are the water flow and uptake of radionuclides in organisms.

Man is exposed via consumption of water, fish, crustaceans, and milk and meat from cattle, which in turn have consumed water and aquatic plants or have grazed on areas irrigated with contaminated water, similar to the lake typical ecosystem.

5.3.3 Coastal area

The coast module deals with the Swedish Baltic Sea coast. The typical ecosystem has two parts so that the calculations can handle both an archipelago and an open coast, with water exchange between the two. The inflow of radionuclides can take place via the archipelago or the open coast. The water in the different parts is assumed to be homogeneously mixed. Important processes include the distribution of nuclides between solution and solid phase, water turnover, uptake in biota and sedimentation and resuspension.

Exposure is assumed to take place via consumption of fish, aquatic plants (algae) and milk and meat from cattle that have drunk the water and eaten aquatic plants.

5.3.4 Agricultural land

The typical ecosystem for agriculture areas deals with a saturated zone with horizontal groundwater flow approximately one metre below the ground surface and two unsaturated zones above the saturated one called ‘deep soil’ and ‘topsoil’. The radionuclides are brought in with the contaminated groundwater and can be transported to the upper soil layer by means of capillary forces, diffusion, plant roots and bioturbation (e.g. soil-living worms). The contaminated soil is used as cropland or pastureland.

Exposure pathways are consumption of cereals, leafy vegetables and root crops, plus milk and meat from animals fed with feed produced on contaminated land. Inhalation of radioactive dust and external exposure from the contaminated soil are also included.

5.3.5 Peat bogs – wetlands

Many dissolved substances have a strong tendency to be taken up and accumulated in peat. This includes most of the radionuclides as well. The peatland deals with radionuclides that reach peat bogs and wetlands via groundwater outflows. The distribution of radionuclides in the peat is assumed to be homogeneous. The peat is used in horticulture or as fuel in heating plants and private households. Important processes are adsorption of radionuclides on peat as well as the processes that take place in the soil when the peat is used as soil conditioner, plus uptake in organisms.

Man is exposed via consumption of milk, meat, cereals, leafy vegetables and root crops, and by inhaling radionuclides present in dust from the peat and in flue gases from peat burning. External exposure from peat is also taken into account.

5.3.6 Well

The well is assumed to contain groundwater contaminated with radionuclides. The well is described in the typical ecosystem as one compartment. The turnover of water in the well is assumed to be the annual capacity that is normal for a well on that particular site. In the well we disregard all sorption on materials present there. The water in the well is used as drinking water for people and cattle and for irrigation of a small vegetable garden where the vegetables that are consumed annually are grown.

Exposure pathways are consumption of drinking water, leafy vegetables, root crops, and milk and meat from cattle that have drunk contaminated water. External exposure from the vegetable garden, as well as inhalation of dust from the same, are also taken into account.

5.3.7 Irrigation

Irrigation is handled as a subroutine in the calculation program that is included in the typical ecosystems for well, running water and lake. The dissolved or particle-bound radionuclides in the water in the typical ecosystem's water compartment are transferred to plants and soil via irrigation. The amount transferred depends on the irrigation volume and the water's radionuclide content. The analysis takes into account retention of nuclides in soil and on plants, transport in the soil by soil organisms, and redistribution to the edible parts of the plants. Precipitation minus evaporation influences the migration of radionuclides in the soil.

Man is exposed by eating cereals, leafy vegetables and root crops, and milk and meat from cattle that have grazed on irrigated areas, but also by inhalation of dust from cultivated areas, as well as by exposure to external radiation from the contaminated land.

5.4 Description and classification of the sites

The three study sites Aberg, Beberg and Ceberg were divided into subareas (250 × 250 m), and each subarea was assigned to a typical ecosystem based on the existing ecosystem type. The peatland module was used for forest, for which a module was lacking. Maps of the areas and data on water exchange and precipitation on the three sites have been used to calculate the ecosystem-specific dose conversion factors, EDF (Nordlinder *et al.*, 1999). Following are brief descriptions of the three sites (Lindborg and Schüldt, 1998).

5.4.1 Aberg

Aberg is a coastal area on the Baltic Sea. The landscape is dominated by archipelago, but also includes open coast. The soil layer is thin and the terrestrial vegetation is dominated by pine forest with rock outcrops.

The typical ecosystem for a coast (archipelago and open coast) is mainly chosen to deal for this area, see Table 5-2. Forestland and peatlands also comprise a large portion of the ecosystems. Small parts of the site are agricultural land, while there are no lakes. There are also three wells in the area, see Table 5-3.

Table 5-2 Site-specific data for coast of Aberg (Nordlinder *et al.*, 1999).

	Mean	Variation (min. – max.)
Archipelago, area	$1.4 \cdot 10^6$ m^2	$1.3 \cdot 10^6 - 1.5 \cdot 10^6$ m^2
Archipelago, mean depth	2.3 m	2.1 – 2.5 m
Archipelago, max. depth	8.0 m	7.5 – 8.5 m
Archipelago, residence time	45 days	42 – 48 days
Open coast, volume	$1.7 \cdot 10^8$ m^3	$1.4 \cdot 10^8 - 2.0 \cdot 10^8$ m^3
Open coast, mean depth	7 m	6 – 8 m
Open coast, no. of turnovers per year	44 yr^{-1}	30 – 57 yr^{-1}

Table 5-3 Well capacity in Aberg (Nordlinder *et al.*, 1999).

Well No.	Capacity	Variation (min. – max.)
1	200 l/hr	180 – 220 l/hr
2	300 l/hr	270 – 330 l/hr
3	400 l/hr	360 – 440 l/hr

The capacity of the average well in the area is 300 l/hr.

5.4.2 Beberg

Beberg is a relatively flat area. The surface layer is dominated by glacial till, peat and rock outcrops. The vegetation consists primarily of forest, but there is also some cultivated land. Wetlands are common due to the flatness of the landscape, but large portions of the forestland and arable land have been drained by ditching (Sundblad and Bergström, 1983).

Forests comprise the dominant ecosystem, followed by wetlands and agricultural land. There are also lakes and watercourses, see Table 5-4. In addition, there are four wells with varying capacity in the model area, see Table 5-5. Since forestland is represented by peatland, the peat typical ecosystem will dominate.

Table 5-4 Site-specific data for lake typical ecosystem in Beberg (Nordlinder *et al.*, 1999).

	Mean	Variation (min. – max.)
Area	4.3 km^2	3.9 – 4.7 km^2
Greatest depth	4.1 m	3.7 – 4.5 m
Catchment area	117 km^2	105 – 129 km^2

Table 5-5 Well capacities in Beberg (Nordlinder *et al.*, 1999).

Well No.	Capacity	Variation (min. – max.)
1	400 l/hr	360 – 440 l/hr
2	700 l/hr	630 – 770 l/hr
3	900 l/hr	810 – 990 l/hr
4	3,000 l/hr	2,700 – 3,300 l/hr

The capacity of the average well in the area is 1,000 l/hr.

5.4.3 Ceberg

Ceberg is situated within a mountainous coastal area in northern Sweden, approximately 20 km inland from the Bothnian Sea coast. The area is relatively flat compared with its surroundings. The elevation varies from 100 to 150 metres above sea level. The vegetation is dominated by pine forest, and there are areas with peat and exposed bedrock (Lindborg and Schüldt, 1998).

The site is drained by two major watercourses, along one of which are small areas of cultivated land. Peatlands and forests are the dominant ecosystems. As a result, peat modules dominate in the area. Typical ecosystems for watercourses (Table 5-6) also occur frequently, as well as agricultural land modules. There are also two wells with different capacities within the area, see Table 5-7.

Table 5-6 Catchment areas for the watercourses in Ceberg (Nordlinder *et al.*, 1999).

	Mean	Variation (min. – max.)
Brook, area	580 km^2	570 – 590 km^2
Rapids, area	3,430 km^2	3,420 – 3,440 km^2

Table 5-7 Well capacities in Ceberg (Nordlinder *et al.*, 1999).

Well No.	Capacity	Variation (min. – max.)
1	400 l/hr	360 – 440 l/hr
2	600 l/hr	540 – 660 l/hr

The capacity of the average well in the area is 500 l/hr.

5.5 Discussion

The highest dose conversion factors in these calculations are obtained with the typical ecosystems peat, well and agricultural land. This is due to the dilution values used, and the properties of the substances. Drinking water is a dominant exposure pathway for actinides in the well. For the more bioavailable nuclides such as chlorine, iodine and cesium isotopes, consumption of foodstuffs leads to higher doses than consumption of water. It is also interesting to note that the typical ecosystem for agricultural land, which pertains to transport of the nuclides from below to the soil, can give doses that exceed those from a well.

Calculations of uncertainty showed that the parameter values used to describe the biological processes such as root uptake, accumulation in fish and secretion of the nuclides to milk and meat made the biggest contribution to uncertainty (Bergström *et al.*, 1999). These were followed by the physical/chemical factors, particularly those that describe sorption, K_d values. The parameter values used to describe consumption, living habits etc. made the least contribution. Unlike other parameters, the ICRP's dose conversion factors have been treated as constants, since information on variation is lacking.

The subdivision of the area into typical ecosystems was done solely according to map studies. Future site investigations will reduce the variation in the input data and permit a better classification of typical ecosystems. The subdivision into 250 × 250 m squares was chosen to suit the calculations, but it can be discussed whether this is optimal. A square may contain several types of biotopes. In such cases, the biotope with the largest area was chosen, but it is possible that there is a greater probability that groundwater discharge takes place to a smaller biotope. A coarser grid could also be used for homogeneous areas.

A rough comparison of the sites from an exposure point of view shows that Ceberg has a potential for higher doses than the others since the peat modules are most frequent there. Aberg has the lowest potential for exposure, since the coast modules provide higher dilution than other surface water modules.

These studies can be viewed as the first step in improving the calculation of doses in safety assessments of geological repositories for radioactive waste. Development of models for radionuclide turnover in forests is under way, along with an analysis of how the methodology can be optimized.

5.6 References

Bergström U, Edlund O, Evans S, Röjder B, 1982. BIOPATH - A computer code for calculation of the turnover of nuclides in the biosphere and the resulting doses to man. STUDSVIK/NW-82/261, Studsvik AB.

Bergström U, Nordlinder S, Aggeryd I, 1999. Models for dose assessments. Modules for various biosphere types. SKB Technical Report TR 99-14, Svensk Kärnbränslehantering AB, Stockholm.

Lindborg T, Schüldt R, 1998. The biosphere at Aberg, Beberg and Ceberg - a description based on literature concerning climate, physical geography, ecology, land use and environment. SKB Technical Report TR 98-20, Svensk Kärnbränslehantering AB, Stockholm.

Nordlinder S, Bergström U, Mathiasson L, 1999. Ecosystem specific dose conversion factors for Aberg, Beberg and Ceberg. SKB Technical Report TR 99-15, Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1999. SR 97 – Main Report. Svensk Kärnbränslehantering AB, Stockholm.

Sundblad B, Bergström U, 1983. Description of recipient areas related to final storage of unprocessed spent nuclear fuel. SKB/KBS Technical Report TR 83-11, Svensk Kärnbränslehantering AB, Stockholm.

6 Description of Reference Scenario

6.1 Introduction

In an earlier prestudy of the barrier functions in the near field of SFL 3-5 (Wiborgh, 1995), the PID (Process Influence Diagram) methodology was used to identify and structure events and processes that can influence the long-term performance of the barriers. A reference scenario was defined by specifying the initial state of the barriers and the conditions in the surrounding environment, i.e. in the far field. The importance of the identified events and processes with their initial and boundary conditions was then assessed on the basis of then-existing knowledge of these processes and events. This resulted in a reference scenario that served as a basis for quantitative analyses of radionuclide release from the near field. Furthermore, the results highlighted a number of remaining questions regarding the importance of identified events and processes.

A re-examination of the earlier material and an updating as far as possible events and processes are concerned, plus their importance for the performance of the barriers and the release of radionuclides from the near field, have been carried out as a part of the preliminary safety assessment. In this updating, processes and events as well as the interaction between them have been structured in the form of THMC (Thermal, Hydrological, Mechanical, Chemical) diagrams. A THMC diagram is actually a table where the first column contains all identified processes and events, sorted under the headings Thermal, Hydrological, Mechanical or Chemical. The headings of the other columns give the parameters needed to describe states and properties of the physical components in the system. These parameters can also be classified as Thermal, Hydrological, Mechanical or Chemical. Arrows in the table then indicate how the different processes are influenced by and themselves influence the various parameters. The principle of these diagrams is described in greater detail in SR 97 (SKB, 1999).

The results of the re-examination of processes and events that can influence the barriers, and the release of radionuclides and other dissolved substances from the near field in SFL 3-5 for a defined reference scenario, are summarized in this chapter. The summary consists of descriptions of the expected evolution of temperature (T), water and gas flows (H), physical/mechanical properties (M) and chemical properties of the barriers, and water composition in the repository (C). The processes that release radionuclides and toxic metals from the waste and transport them in the repository's near field are then described. These descriptions serve as a basis for assumptions and choice of data for the calculations of the release of radionuclides and toxic metals from the near field of SFL 3-5 that are presented in Chapter 8.

6.2 Premises for the Reference Scenario

The description of the expected evolution of the repository's near field apply for a reference scenario. An important premise for the reference scenario is a stable far field around the repository, which means that the thermal, hydrological, mechanical and chemical conditions in the rock surrounding the repository are assumed to remain unchanged over time. Another premise is that all barriers in the near field perform as expected at the time the repository is closed.

6.3 Thermal Evolution

When SFL 3-5 is closed, the temperature in the different parts of the repository and in the host rock may be slightly higher than is normal at that depth. The higher temperature is caused by exothermic (heat-generating) processes in the repository and ventilation during the operating period. Examples of such exothermic processes are decay of radionuclides in the waste and hydration of the concrete in the various enclosures in the repository.

After closure, radioactive decay in the waste is the only process that generates heat. The heat is transported away from the waste by heat conduction in the waste packages and surrounding barriers. Calculations show that heat evolution in the waste leads to an increase in the mean temperature in the SFL 5 tunnel and 20 m of the surrounding rock by about 3ºC during the first 100 years after closure (Skagius *et al.*, 1999a). This applies regardless of whether the repository is water-saturated or not during the first 100 years. Heat transport is rapid, which means that the difference between the temperature in the waste and the calculated mean temperature is negligible. The calculation presumes that all radiation energy is converted to heat inside the waste packages in SFL 5. This is a reasonable assumption. An estimate of the radiation attenuation in a waste package in SFL 5 shows that at least 99.6% of the radiation energy from the waste is absorbed in the waste package (Lindgren *et al.*, 1998).

Heat is also generated in SFL 3 and in SFL 4. But the heat that is generated in these two repository parts is only 10% of the heat generated in SFL 5, so the temperature impact is negligible.

6.3.1 Handling in the safety assessment

The process of radioactive decay in the waste generates heat that causes a temperature increase in the repository’s near field of less than 5ºC during the first 100 years. We assume that this small temperature increase does not influence processes that are of importance for the performance of the near-field barriers, nor does it influence the transport of radionuclides or other dissolved substances from the repository.

6.4 Water flows

6.4.1 Saturation phase

Water transport into the SFL 3-5 repository with its tunnels and near-field rock is determined primarily by the regional water flow in the area. During the construction and operating phase, the repository will be kept dry by the pumping-out of seeping groundwater. This will give rise to a significant reduction of the natural groundwater pressures in the rock around the rock chambers.

When the repository is closed and pumping ceases, groundwater will run into the rock vaults from all directions due to the lower pressure in the rock vaults and in the surrounding rock. When the different parts of a rock vault have been filled with water, the water pressure in the vault will increase. Eventually a stable pressure and a steady-state water flow through the repository will be reached. The water flow into the repository area and the principal direction of water flow in the repository area will then be determined by the regional groundwater flow.

The time it takes to reach steady-state conditions depends on the groundwater's inflow rate, which can vary as much as 0.3–300 m^3 per length metre of tunnel and year, according to experience from SFR, the Stripa Mine and the Äspö HRL. The restoration time can therefore vary from a few tens of years to several hundred years, depending on the hydraulic properties of the surrounding rock (Karlsson *et al.*, 1999). Saturation proceeds faster in a generally more conductive rock with larger water-bearing zones than in a less conductive rock.

6.4.2 Long-term changes

In the long-term perspective, the water flow into the repository area and the principal direction of water flow in the repository area are determined by the regional groundwater flow. Since the reference scenario assumes that the regional groundwater conditions are stable, no changes occur in either the direction or the size of the groundwater flow into the repository area.

Owing to the layout of the repository, with the SFL 4 tunnel encircling the two rock vaults SFL 3 and SFL 5, the principal throughflow of water in the repository area will take place in SFL 4. How much water flows through the rock vaults and how this water flow is distributed within each rock vault depends on how water-permeable the engineered barriers in the repository are, both in relation to each other and in relation to the surrounding rock.

Cracking and clogging of pores in the barrier materials can eventually alter the water permeability of the barriers and thereby also alter the magnitude of the water flow in different parts of the repository. Cracking may occur as a result of gas pressure build-up inside the repository. Chemical reactions between infiltrating water and barrier materials can lead to precipitation and clogging of pores in the materials. Processes that may lead to changes in the long-term properties of the barriers, including water permeability, are described in greater detail in Section 6.7.1.

Gas forms inside the repository, see Section 6.5. The expulsion of water from concrete and cement that occurs when the gas creates transport pathways through the materials can have a short-lived effect on water flows in the materials. Once gas channels have been created, the influence on the water flows in the barrier materials is expected to be negligible. The reason for this is that only a few gas channels through the barriers are sufficient for the gas to escape from the repository at moderate gas pressures (Skagius *et al.*, 1999b).

Settlements may occur in the gravel fill in the SFL 4 tunnel and in the SFL 3 and SFL 5 rock vaults already during the water saturation phase, but also in a longer-term perspective. A zone with high water permeability may thereby be created next to the roof in the SFL 4 tunnel and the rock vaults. This is not expected to influence the magnitude of the water flow into the SFL 4 tunnel or the rock vaults, but may have some influence on the distribution of the water flow within a given repository part.

6.4.3 Handling in the safety assessment

A generic hydrological model of the near field has been used to calculate the water flow through the different barriers in the near field (Holmén, 1997). The calculations are based on various assumptions concerning contrasts between the conductivity of the near-field barriers and the surrounding rock mass. Furthermore, the calculations assume a water-saturated repository and steady-state hydrological conditions in surrounding rock. The influence of gas formation and gas transport inside the repository on the water flows in the different repository parts is assumed to be negligible. Modelling of the water flow in the near field is described in greater detail in Section 7.3.

The water flows in the near-field barriers that have been used in the calculations of transport of radionuclides and toxic metals are presented in Section 7.5. The assumed water flows are based on the results of the hydrology modelling, under the assumption that the near-field barriers have hydraulic conductivities that can be expected after long times.

6.5 Gas

6.5.1 Gas formation

Air trapped inside the repository at closure is compressed by the intruding water. According to Boyle's law, a hydraulic pressure of 30 bar at a depth of 300 m will reduce the air volume by a factor of 30. The air will dissolve in the water and the oxygen will be consumed by e.g. corrosion of steel, see further Section 6.6.

Once the oxygen has been consumed, anaerobic steel corrosion can begin. This process generates hydrogen, which can contribute to increased gas pressure in the repository. Steel is present in the waste and in waste containers in all three repository parts, as well as in reinforcement bars in the concrete enclosures in SFL 3 and SFL 5.

The waste in SFL 3 also contains aluminium and organic materials. Aluminium is relatively stable in groundwater, but corrodes rapidly at elevated pH, forming hydrogen. Certain bacteria can be active in the absence of oxygen. Anaerobic biodegradation of organic materials such as cellulose is therefore a possible source of methane and carbon dioxide. Carbon dioxide dissolves in the water and may be consumed by reactions in the concrete, but the methane can contribute to increased gas pressure.

Gas formation rates and the total quantity of gas that can form in the three repository parts have been calculated (Skagius *et al.*, 1999b). Gas formation in an SFL 3 repository situated at a depth of 500 m has been calculated to be about 450 m^3/y (equivalent to about 750 m^3/y at a depth of 300 m). This is in a case where corrosion of both aluminium and steel in the repository contributes to hydrogen gas formation, while microbial degradation of organic materials generates gaseous methane. This gas formation rate is dominated completely by the contribution made by aluminium corrosion, which is rapid. When all aluminium has corroded away (after about 5 years at an assumed corrosion rate of 1 mm/y), the gas formation rate declines to around 20 m^3/y (about 30 m^3/y at a depth of 300 m), more than half of which comes from microbial degradation of organic material in the waste. After about 600 years, all organic material has been consumed, and after about 8,000 years all steel, in the repository.

In SFL 4, gas is formed solely by anaerobic corrosion of steel. The gas formation rate has been calculated to be about 25 m^3/y for a repository situated at a depth of 500 m (about 40 m^3/y at a depth of 300 m) (Skagius *et al.*, 1999b). As steel components of differing thicknesses are consumed, gas formation in the repository abates. Corrosion of the thickest components will, however, contribute to gas formation for about 150,000 years.

Gas formation in SFL 5 in connection with anaerobic corrosion of steel has been calculated to be about 8 m^3/y for a repository situated at a depth of 500 m (about 13 m^3/y at a depth of 300 m) (Skagius *et al.*, 1999b). The waste in SFL 5 also contains Zircaloy, which can give rise to hydrogen by corrosion of zirconium. Due to expected low corrosion rates, however, the contribution from corrosion of Zircaloy is negligibly small (Skagius *et al.*, 1999b).

Radiation from the radioactive waste can cause decomposition of water inside the waste packages. However, an earlier study has shown that this process gives rise to negligibly small amounts of gas compared with the other gas-forming processes (Wiborgh, 1995).

6.5.2 Gas transport

Gas transport out of the repository is driven by differences in gas pressure between the different barriers. Cement and concrete contain capillary-bound water, where the capillary pressure declines with increasing pore size and crack width. This means that an internal gas over-pressure is required before gas passages form in packages containing these materials. If the concrete or the cement does not contain any penetrating cracks from the start, the built-up gas pressure may create such cracks and the gas will then be transported out through these newly-formed cracks.

The porous concrete surrounding the waste packages in SFL 3 and SFL 5 has a relatively high permeability to gas. This means that it does not require particularly high gas pressures for the gas to create transport pathways through the porous concrete. The gas will probably flow upward through the backfilled concrete, forcing the water out into the transport pathways. If the lid on the concrete enclosure is not completely watertight, gas and the water expelled by the gas can be transported out through these gaps. If the lid is tight and there are no other penetrating cracks or gaps in the concrete enclosure, a gas pressure will be built up in the enclosure. If penetrating cracks or gaps eventually form in the concrete structure, for example due to a build-up of gas pressure, the gas can flow out through them and the gas pressure inside the concrete structure will decrease.

Gas can also form inside the walls, bottom and lid of the concrete structure as a result of corrosion of reinforcement bars. This gas will presumably find its way out through cracks adjacent to the reinforcement bars to the porous concrete inside, or the gravel fill outside, the concrete structure. If such cracks do not exist when the repository is closed, they will probably form with time due either to the pressure built up by the gas itself or the mechanical pressure exerted by the expanding corrosion products on the surrounding concrete, see Section 6.7.

Once gas-conducting passages have been formed in the different barriers in the repository, gas will flow out through them as long as the pressure difference exceeds the capillary pressure there. The capillary pressure in an intact structural concrete is of the order of 1–2 MPa (Moreno and Neretnieks, 1991), while the capillary pressure in a plane-parallel crack with an aperture of 0.1 mm is of the order of 1–2 kPa (Skagius *et al.*, 1999b). Thus, a gas pressure of 1–2 kPa is required in order to create gas passages in such cracks in the concrete enclosure in SFL 3 and SFL 5. To then transport out the gas formed in the waste packages and concrete structure in SFL 3 and SFL 5 without any further increase in the internal gas pressure, it is sufficient if the total cross-sectional area of the crack aperture is of the order of 10^{-4} m^2 (Skagius *et al.*, 1999b). This is equivalent to, for example, one penetrating crack that is 1 m wide with a crack aperture of 0.1 mm or ten cracks each of which is 0.1 m wide and has a crack aperture of 0.1 mm.

The capillary forces in the gravel fill in the different repository parts are small, and only small positive gas pressures are required to open up transport pathways for the gas and to drive the gas through the porosity in the fill (Skagius *et al.*, 1999b). With time, the gravel fill in SFL 3 and SFL 5 may be affected chemically by leached-out concrete pore water, which may slightly reduce the porosity of the fill, see Section 6.7. However, this is not expected to influence gas transport in the fill.

The gas that flows upward through the gravel fill eventually reaches the void between the fill and the roof in the SFL 4 tunnel and the rock vaults before it passes out through fractures in the rock. Since the gas can spread through the entire void under the roof, all fractures that intersect the roof will be available for gas transport. Studies that have been made suggest that the fracture systems that are normally present in the rock has a large capacity for gas transport (Thunvik and Braester, 1986; Wikramaratna *et al.*, 1993).

6.5.3 Handling in the safety assessment

The calculations that have been done indicate that a few cracks or gaps in concrete barriers in the repository are sufficient for the gas that forms inside the repository to escape at even moderate gas pressures. These cracks or gaps will either be present from the start or they will be created by the gas pressure that builds up over time. This is taken into account in the safety assessment in the evaluation of the long-range properties of the concrete barriers, see Section 6.7.3.

Formation and transport of gaseous radionuclides is dealt with in the safety assessment. Release of radioactive gas is further described in Section 6.9.

Direct effects of gas formation and gas transport on water flows in the repository are neglected in the safety assessment. The gas only needs to utilize a small fraction of the barriers' permeability to migrate out of the repository, and the effect on the water flow should therefore be small. Any internal gas pressure could possibly counteract water inflow, which would be a positive effect.

6.6 Hydrochemical conditions

The redox potential, pH and salinity of the water, and its content of complexing agents and colloids, are of great importance for the performance of the near-field barriers and for the migration of radionuclides and other solutes. The groundwater that enters the repository after closure will be affected by the materials in the repository via various processes. The most important processes and their influence on the composition of the water in the near field are summarized in this section. More detailed descriptions can be found in Karlsson *et al.* (1999) and Skagius *et al.* (1999a).

6.6.1 Influencing processes

The downflow of surface water decreases when the groundwater level in the repository is restored. If a heavier saline groundwater lies beneath the repository, its upflow will decrease as soon as the repository drainage work (pumping and ventilation) ceases. This does not necessarily lead to a complete restoration of the groundwater composition that existed prior to all interventions in the hydrology of the area. A large proportion of water from the surface will probably remain and be mixed with more saline water in a diffuse mixing zone. But no major deviations from the original composition are expected.

The groundwater that enters the repository after closure will be affected by the materials in the repository, particularly steel and concrete. Oxygen in trapped air will dissolve in the water and react and be consumed by e.g. corrosion of steel, oxidation of dissolved iron(II) and sulphide and bacterial degradation of organic material. When the oxygen has been used up, anaerobic steel corrosion can follow.

Anaerobic steel corrosion

Anaerobic corrosion of steel produces iron compounds (corrosion products) and Fe^{2+} ions that are released to the water. These ions are also released from the corrosion products and therefore continue to keep the redox potential down after all steel has corroded away. The hydrogen formed by corrosion can in principle also act as a reducing agent, but this requires redox processes where hydrogen is active at normal temperatures.

Leaching of concrete

Reinforced concrete will be used to build SFL 3 and SFL 5, and porous concrete will be used as backfill between the waste containers. Some of the containers will be made of reinforced concrete, and in certain cases the waste will be stabilized with cement or be immobilized in concrete. The composition of the cement will correspond to that of Degerhamn Standard Portland Cement, see Section 3.4.1.

The concrete particularly affects the pH of the intruding groundwater. Fresh concrete can cause a pH that is higher than 13 due to leaching of readily soluble alkali hydroxides. When they are gone, the pH is buffered to about 12.5 by calcium hydroxide (portlandite), which has been demonstrated in cement leaching tests, see Section 3.4.1

and Table 3-2. If the portlandite is also leached out and disappears, a dissolution of calcium silicate hydrate begins. A gradual reduction of pH to about 10 occurs during this dissolution phase, see Section 3.4.1.

The components that are leached from the concrete will affect the composition of the intruding water. The sodium content of the intruding groundwater can increase if it is poor in sodium compared with concrete pore water. The potassium content is, on the other hand, always lower in groundwater than in concrete pore water, so the potassium concentration will increase. The calcium concentration at high pHs will be controlled by the solubility of portlandite. At high pH, any magnesium in the intruding water will be precipitated as brucite ($Mg(OH)_2$). A high calcium concentration means that carbonate ions will be precipitated as calcite ($CaCO_3$).

Sulphate and to some extent chloride can be taken up by cement. Leaching tests have however shown that the concentrations will remain relatively high (Engkvist *et al.*, 1996).

Degradation of organic materials

The organic waste that is present in SFL 3 will be affected by the intruding groundwater. A high pH can cause hydrolysis of organic materials. Hydrolysis of cellulose is particularly important, since this process yields products that can form complexes with radionuclides (Allard *et al.*, 1995).

Experimental studies of alkaline degradation of cellulose have shown that the principal degradation product that can form complexes with radionuclides is isosaccharinic acid, ISA (Greenfield *et al.*, 1995; Pavasars, 1999). ISA can thereby influence the nuclides' solubility and sorption on solid surfaces. The solubility of actinides can increase substantially in water with a high pH when cellulose is present (Bradshaw *et al.*, 1986 and 1987; Greenfield *et al.*, 1992).

Bacterial degradation of the organic waste in SFL 3 is also conceivable, even though it is not an ideal environment for microorganisms in view of the high pH. In the presence of oxygen there is a theoretical possibility that bacteria can form ferrocenes, which are strong complexing agents. In the absence of oxygen, the degradation products are usually carboxylic acids, which are not particularly strong complexing agents. Bacteria can also contribute towards degrading ISA that is formed by alkaline hydrolysis of cellulose (Lessart *et al.*, 1996).

Colloids – formation and stability

Sufficiently small particles (smaller than 1 μm) of solid substances can occur as colloids in the groundwater. Analyses have shown that Swedish groundwaters contain particles of mud, silica and iron(III) hydroxides. Even if surface waters such as lakes and rivers are rich in particles, groundwaters (and particularly deep groundwaters) are very poor in colloids. The measurements that have been done indicate concentrations around 20-45 μg/l (Laaksoharju *et al.*, 1995). The reason for these low concentrations is that deep groundwaters are strongly mineralized, i.e. contain a lot of dissolved salts. Electrolytes in the groundwater destabilize colloids so that only very low concentrations are possible. The electrolyte is said to coagulate the colloid (Hunter, 1987). The mineral particles are generally negatively charged, so the positive ions in the groundwater are of the greatest importance for the stability of the colloids. Positive ions in the water reduce the repulsion between negatively charged particles so that they can be bound to each other. The positive ions have a greater effect the higher their charge is (strongly polarizing ions). Trivalent ions such as Fe^{3+} and Al^{3+} are unfortunately present in much too low concentrations in the groundwater to be of importance, but there are plenty of divalent ions such as Ca^{2+}. Ideally, the concentration of the calcium ion Ca^{2+} in the water should be higher than 10^{-4} mol/l (4 mg/l) (Laaksoharju *et al.*, 1995). This condition is met in the deep groundwaters, which not seldom exhibit considerably higher levels. Other ions such as Na^+, K^+ and Mg^{2+} contribute towards keeping down the colloid concentrations.

The cement could also give rise to particles, but it also contributes electrolytes, which keep down the colloid concentrations. An example is found in the studies of the natural analogue to cement in Maqarin in Jordan (Smellie *et al.*, 1999). Here, cement minerals have been formed on several occasions by self-ignition and fire in the bituminous marl that is present in the area. As a result, the groundwater at Maqarin has a pH of about 12.5 and a composition that in other respects as well resembles leachate from old concrete. Particles in the groundwater have been analyzed, but the concentrations are very low and do not differ from what is normal in deep groundwaters.

Reactions between leaching products from concrete and surrounding gravel fill

Alkali hydroxides and portlandite ($Ca(OH)_2$), which are leached from the concrete in enclosures and waste packages, can react chemically with the gravel fill outside. Primary silicate minerals such as quartz and feldspar are dissolved, and secondary calcium silicate hydrate (CSH) phases are precipitated at the same time as hydroxide ions are consumed (Savage, 1997; Holgersson *et al.*, 1998b).

The reactions take place for the most part on the surface of the solid materials. Analyses of rock specimens exposed to cement pore water show surface coatings of secondary phases that are roughly 50 to 100 µm thick and reaction depths that are less than 50 µm (Holgersson *et al.*, 1998b). Analysed changes in the concentration of calcium, silicon, sodium and potassium in a reacted cement water indicate, however, that the rate-determining step is dissolution of silicate minerals, and that the reaction is not limited by diffusion through the secondary phases on the surface (Savage *et al.*, 1992). Diffusion tests also show that sodium ions can diffuse through secondary phases formed on the surface of rock specimens exposed to cement water (Holgersson *et al.*, 1998b).

6.6.2 Expected water composition

The expected composition of the water in the different repository parts is given in Table 6-1 for a saline and a non-saline intruding groundwater (reference waters for Aberg and Ceberg, see Table 4-4. For SFL 3 and 5, these water compositions are expected to exist of the order of a hundred thousand years after repository closure. This, however, assumes that no changes occur in the composition of the incoming groundwater.

Table 6-1 Expected water composition in the near-field barriers in SFL 3-5 for a saline versus a non-saline intruding groundwater.

	Concrete enclosure, SFL 3 and 5 and waste packages, SFL 4		Gravel fill, SFL 3, 4 and 5	
	Saline water	**Non-saline water**	**Saline water**	**Non-saline water**
Redox	Reducing conditions		Reducing conditions	
pH	13.1 → 12.5[a]	13.1 → 12.5[a]	7.7 – 12.5[b]	9.3 – 12.5[b]
Important cations				
Na^+ (mmol/l)	91	28 → 5[a]	91	5
K^+ (mmol/l)	83 → 0.2[a]	83 → 0.05[a]	0.2	0.05
Ca^{2+} (mmol/l)	1 → 20[a]	1 → 20[a]	≤ 47	≤ 0.5
Mg^{2+} (mmol/l)	≤ 0.02	≤ 0.02	≤ 2	≤ 0.04
Important anions				
Cl^- (mmol/l)	181	5	181	5
$CO_3{}^{2-}{}_{tot}$ (mmol/l)	0.1	0.1	≤ 0.2	≤ 0.3
$SO_4{}^{2-}$ (mmol/l)	6	0.001	6	0.001
Complexing agents				
ISA (mmol/l)	≤ 0.1[c]	≤ 0.1[c]		
Colloids (mg/l)	< 0.05	< 0.05	< 0.05	< 0.05

[a] The arrow → shows how the water will change with time

[b] Possible range of pH in the water in the gravel fill

[c] inside waste packages with cellulose in SFL 3

Redox conditions

The oxygen present in the repository at closure is rapidly consumed by corrosion of steel, oxidation of dissolved iron(II) and sulphide, and microbial activity. The Fe^{2+} ions that are released during anaerobic steel corrosion and from steel corrosion products keep the redox potential low in all repository parts.

pH

Groundwater intruding into the repository after closure is affected by the concrete. The pH inside the concrete enclosure in SFL 3 and 5 and inside the waste packages in SFL 4 is therefore expected to be the same as in pore water in fresh cement, see Table 3-2. Leaching of the alkali hydroxides and portlandite in the concrete gradually leads to a reduction of the pH to about 12.5. The time it takes to leach the portlandite out of the concrete in SFL 3 and SFL 5 has been estimated to be of the order of a hundred thousand years and possibly longer (Karlsson *et al.*, 1999). This means that the pH inside the concrete enclosure in SFL 3 and SFL 5 will not fall below about 12.5 during this time. Due to the greater water turnover in SFL 4 and a shorter diffusion distance for leached-out components inside the waste packages in SFL 4, the portlandite will be leached out faster than in the concrete enclosures in SFL 3 and 5. The time it takes to leach out all portlandite from the concrete in waste packages in SFL 4 is nevertheless expected to be very long.

Leaching of alkali hydroxides and portlandite out of the concrete will raise the pH in the gravel fill surrounding the concrete enclosures in SFL 3 and 5 and the waste packages in SFL 4. Reactions with silicate minerals in the gravel fill are, however, expected to neutralize the pH. Simple mass balance calculations show that neutralization of all hydroxide from the concrete in SFL 3 and 5 is possible without exceeding a reaction depth of 50 μm in the fill material (Karlsson *et al.*, 1999). Other calculations indicate that the kinetics for silicate dissolution in the fill materials are faster than the rate at which hydroxide ions are leached from the concrete (Karlsson *et al.*, 1999). Based on these results, the pH in the rock outside the gravel fill is not expected to exceed the pH of the groundwater. In contrast, the water in the gravel fill in all repository parts can vary between the pH in incoming groundwater and the pH in concrete water, depending on time and distance from the concrete.

Important cations

The concentration of sodium is higher in concrete water than in a non-saline incoming groundwater. During a short period, the concentration of sodium in the concrete is therefore determined by the sodium leached from the concrete when incoming groundwater is non-saline. Over the longer term, and with a saline incoming groundwater, the concentration of sodium in the concrete will be the same as in the groundwater.

After closure of the repository, the concentration of potassium in the concrete is determined by the alkali hydroxides in the cement. Leaching of the alkali hydroxides leads to a lowering of the potassium concentration. When the alkali hydroxides have been leached out, the concentration of potassium in the concrete is the same as in the groundwater.

The concentration of calcium in the water in the concrete is determined by the solubility of portlandite and the water's pH. The magnesium concentration is determined by the solubility of brucite ($Mg(OH)_2$).

Influx of potassium from the concrete is not expected to appreciably affect the potassium concentration in the gravel fill, due to dilution and water flux in the gravel fill. The calcium concentration is assumed to be the same as in the groundwater. Some depletion of calcium may, however, take place in areas with elevated pH due to formation of CSH phases and precipitation of calcite.

Important anions

The concentrations of chloride and sulphate in the water in the concrete are primarily determined by their concentrations in the intruding groundwater. Sulphate and chloride can be taken up by cement, but this uptake is small and is not expected to significantly affect the concentrations. The carbonate concentration in the concrete water is determined by the solubility of calcite and the concentration of calcium ions.

In the gravel fill, the chloride and sulphate concentrations are determined by the composition of the groundwater. The concentration of carbonate-bicarbonate is assumed to be the same as in the groundwater. Some depletion of dissolved carbonate can, however, take place in areas with elevated pH due to precipitation of calcite.

Complexing agents

Certain waste types in SFL 3 contain cellulose, which can degrade under alkaline conditions to isosaccharinic acid, ISA. The concentration of ISA in these waste packages has been calculated from estimated cellulose quantities, experimentally determined yields of ISA on degradation of cellulose, and measured adsorption of ISA on cement (Skagius *et al.*, 1999a).

The waste and the near-field barriers are not expected to contribute with any inorganic complexing agents of importance for the solubility and sorption of radionuclides. One waste type in SFL 3 does contain cyanides in the form of ferrocyanide used as a precipitant for Cs. This compound can readily dissolve at high pHs. The cyanide enters into solution as the anion hexacyanoferrate, but this anion forms only weak complexes with metallic cations (Bradbury and Van Loon, 1998, page 43).

Colloids

We have not systematically examined all possible causes of particle formation when, for example, pH changes. This lies beyond the scope of this study, which is merely a preliminary safety assessment. But by reason of the results from Maqarin (see "Colloids – formation and stability" in Section 6.6.1) and the fact that the composition of the groundwater ultimately controls the stability of the colloids, we draw the conclusion that SFL 3-5 will not cause any increase in the colloid concentration beyond what can normally occur in groundwater at a depth of 300 metres on the investigated sites.

6.6.3 Handling in the safety assessment

The expected composition of the water in the near field in SFL 3-5 (Table 6-1) has served as a basis for the choice of solubilities and sorption constants for radionuclides and toxic metals in the near field. This is described in greater detail in Section 6.8 and in Chapter 8.

6.7 Properties of the barriers

The permeability of the waste packages and other near-field barriers to water and gas is of importance to the migration of radionuclides and toxic metals from the repository. Permeability is mainly dependent on geometries and on porosity and cracking in the materials. A summary of processes that could eventually alter these properties and the material composition of the barriers is given in this section. The composition of the materials may have some bearing on sorption of radionuclides and toxic metals. In most cases, however, the water composition is of greater importance for sorption.

6.7.1 Influencing processes

Build-up of water pressure

After closure, incoming water will fill up the voids inside the repository and increase the water pressure. Watertight waste containers may then be subjected to large pressure gradients – 30 bar (3 MPa) – if the repository is situated at a depth of 300 m. This could lead to deformation and cracking of the waste containers.

Corrosion of steel

There are steel waste containers in the repository. Even if the containers are watertight at repository closure, corrosion of the steel will eventually make them permeable to water and gas.

Besides steel waste containers, steel is also present in the waste in all three repository parts and in the reinforcement in concrete moulds and concrete enclosures in SFL 3 and SFL 5. The corrosion products that are formed by steel corrosion have a greater molar volume than the steel. This could lead to cracking in adjoining materials. In most cases, however, the available expansion volume is sufficiently great, with the exception of reinforcement bars in concrete moulds and concrete structures. The concrete adjacent to the reinforcement bars may therefore crack.

Via sorption and coprecipitation, the corrosion products formed by steel corrosion can also influence the concentration of other metal ions in the water, such as nickel, see further Section 6.8.

Build-up of gas pressure

Corrosion of steel and aluminium and microbial degradation of organic materials form gas in the repository. Gas formation and release from the different repository parts have been calculated (Skagius *et al.*, 1999b). The gas formed in SFL 3 can give rise to such high internal gas pressures that concrete moulds crack after just a few tens of years and cracks form in the concrete enclosure after a few hundred years. The same applies to gas formation in SFL 5, but after longer times – a few hundred years for concrete moulds and up to a thousand years for the concrete enclosure. If the moulds and the concrete structures are not completely gastight from the start – there are, for example, small gaps between walls and lids in moulds and in the concrete structure – the gas can escape from SFL 3 and 5 without high internal gas pressures being built up. In that case, the gas is not expected to give rise to cracking of the concrete. The gas that forms in SFL 4 can escape into surrounding rock without affecting the properties of the barriers.

Leaching of concrete

Leaching of cement components in the concrete can affect the porosity of the concrete. The porosity of fully hydrated structural concrete is about 15%. Calculations show that the porosity after leaching of alkali hydroxides and all portlandite may be higher, about 16% (Höglund and Bengtsson, 1991; Höglund, 1993). After leaching of all calcium silicate hydrate (CSH), the porosity of a structural concrete has roughly doubled, i.e. increased from about 15 to 30% (Höglund and Bengtsson, 1991, and Andersson and Allard, 1986).

Leaching of the concrete can also affect its mechanical strength (Höglund and Bengtsson, 1991). Strength is primarily dependent on the calcium silicate hydrate in the concrete. Leaching of alkali hydroxides and portlandite will therefore not affect strength. The strength is not expected to be degraded until leaching of CSH takes place.

Formation of ettringite and Friedel's salt

Sulphate in water can react with calcium aluminate minerals in cement and form the minerals ettringite or monosulphate hydrate. Monosulphate hydrate is stable in an area with low sulphate concentration, while ettringite is stable at higher sulphate concentrations (Höglund, 1992). Significant amounts of crystal water are bound to the solid phase when ettringite or monosulphate is formed, so the volume increases appreciably. If the expansion is greater than the available spaces in the form of pores and the like in the concrete, this may lead to cracking. Aluminium is needed to form ettringite. Degerhamn Standard Portland Cement is a sulphate-resistant cement with a relatively low aluminium content.

Chloride in water at high concentrations (more than 0.1–0.5 M) can react with calcium aluminate hydrate in the cement and form Friedel's salt (calcium chloroaluminate hydrates). These can in principle give rise to volume expansion just as ettringite (Höglund and Bengtsson, 1991). The formation of Friedel's salt and ettringite compete for the same calcium aluminium hydrate in the cement, so in principle the formation of the one phase could counteract the formation of the other.

Reactions between leaching products from concrete and surrounding gravel fill

Secondary phases of calcium silicate hydrate can precipitate on the surface of the gravel backfill in the repository on reaction with hydroxide ions leached from the concrete, see Section 6.6.1. These phases change the properties of the surfaces, including the specific surface area. Experiments have shown an increase from about 0.3 m^2/g to about 1.1 m^2/g for a rock specimen that has reacted with cement water (Holgersson *et al.*, 1998b). Changes in the properties of the surface can affect sorption of radionuclides and toxic pollutants. In the case of metal ions, sorption on CSH phases is often higher than on rock minerals (Savage, 1997).

The transformation to secondary CSH phases may entail some volume increase in the solid phase and thereby a reduction in the porosity of the gravel fill. This has been shown in column experiments where cement water has been allowed to flow through finely crushed material (Bateman *et al.*, 1995).

Precipitation of calcite and brucite

Magnesium and carbonate in the groundwater that penetrates into concrete in the repository lead to precipitation of brucite ($Mg(OH)_2$) and calcite ($CaCO_3$) in pores and cracks in the concrete. Leaching of alkali hydroxides and portlandite from the concrete can also lead to precipitation of calcite and brucite in the gravel fill outside the concrete. In both the concrete and the gravel fill, these reactions would lead to a reduction in the porosity of the material and thereby in its permeability to water.

Via coprecipitation, these precipitation reactions may also influence the concentration of radionuclides and toxic metals in the water, see further "Precipitation/coprecipitation with calcite" in Section 6.8.2.

Microbial activity

Bacteria can form so-called "biofilms" on surfaces exposed to alkaline water in a repository. This can clog pores on the surface of, for example, a concrete block. Gas expulsion can thereby be prevented, but experiments show that the film bursts at a pressure of only 2 bar (Lessart *et al.*, 1996). The biofilm is a gel and should therefore not be diffusion-tight and prevent diffusion in the concrete pores. Biofilms can also bind metal ions to the surface via sorption and formation of secondary minerals (Brown and Sheriff, 1998).

6.7.2 Expected barrier properties

Waste packages

Waste containers of steel may be leaky already at repository closure and thereby permeable to water. If they are watertight, there is a great risk that they will be deformed and burst when the water pressure in the repository rises after closure. Corrosion will then increase the permeability of the containers due to the fact that the corrosion products that form are more permeable than the original material. However, it takes of the order of 500 years before the thinnest containers are completely corroded through and several thousand years for the thickest (Skagius *et al.*, 1999b).

At repository closure, concrete moulds have low permeability to water. Precipitation of calcite and brucite in the pores after repository closure can make the concrete even less permeable. Some cracking of the concrete adjacent to the reinforcement bars can be expected due to corrosion and formation of corrosion products. Leaching of the concrete is so slow that it does not affect the porosity of the concrete. However, there is a remote possibility that gas generation inside the waste packages will lead to such high internal gas pressures that penetrating cracks form in the concrete after just a few tens to hundreds of years.

Concrete enclosure in SFL 3 and SFL 5

The concrete enclosure in SFL 3 and SFL 5 is expected to be affected in the same way as the concrete moulds. Leaching of the concrete is so slow that it does not affect the porosity of the concrete, but precipitations can clog up pores. Corrosion of reinforcement bars can lead to crack formation around the reinforcement bars. Sulphate in the groundwater could possibly lead to ettringite formation and some cracking of the outer parts of the enclosure (SKB, 1993). The possibility cannot be ruled out that penetrating cracks could form in the concrete due to high internal gas pressures. This could happen during the first 200 years in SFL 3 and during the first 1,000 years after closure in SFL 5 (Skagius *et al.*, 1999b).

Gravel fill

The porosity of the gravel fill may decrease with time due to formation of secondary minerals. Calcite and brucite may precipitate in the void and secondary CSH phases that form on the surface of the solid material may lead to some volume increase in the solid phase and thereby reduced porosity. These secondary minerals form in a zone next to the concrete enclosure in SFL 3 and 5 and next to the concrete-filled waste packages in SFL 4. With time, this reaction zone may spread further out in the gravel fill as components are leached out of the concrete. However, this would take a very long time, since leaching of portlandite out of concrete is a slow process.

The transformation to CSH phases may also lead to changes in sorption properties in the gravel fill, since the specific surface area increases.

6.7.3 Handling in the safety assessment

It is assumed in the safety assessment that all waste containers of steel are already fully permeable to water at repository closure. It is assumed that the corrosion products that eventually form will not affect water flows through containers or diffusive transport of dissolved components in the water. The ability of the corrosion products to retard the outward transport of radionuclides and toxic metals via sorption and coprecipitation is not taken into account.

A certain percentage of the concrete moulds and concrete enclosures in SFL 3 and SFL 5 are assumed to have penetrating cracks already at repository closure. In the calculations of the water flow through the concrete enclosures, hydraulic conductivity in the enclosure and its interior has been set at 10^{-8} m/s. This is equivalent to penetrating cracks every metre to every ten metres, where the fracture aperture is of the order of 10 to 100 μm (Höglund and Bengtsson, 1991). Intact structural concrete has a hydraulic conductivity of the order of 10^{-12}–10^{-11} m/s (Höglund and Bengtsson, 1991). Possible changes in the porosity of the concrete over time due to chemical precipitation are not taken into consideration; instead, typical values for structural concrete and more porous backfill concrete are used in the safety assessment.

The gravel fill in all repository parts is assumed to have a hydraulic conductivity of the order of 10^{-5}–10^{-4} m/s, a porosity of 30% and sorption properties typical for Swedish rock. The small changes in porosity and sorption properties that may occur over very long times are neglected in the safety assessment.

6.8 Dissolution and transport of radionuclides and toxic metals

6.8.1 Dissolution

Dissolution of radionuclides and toxic metals in the waste is dependent on how accessible these substances are to the water that comes into contact with the waste. The composition of the water also affects the solubility of the substance. The waste in SFL 3 is a mixture of different types of materials, where it is difficult in many cases to determine how accessible radionuclides and other substances are for immediate dissolution in water. In SFL 4 and SFL 5, the waste consists of metal components and in SFL 4 of some concrete as well. The surfaces of these waste types are contaminated, which means that the radionuclides are readily accessible for dissolution. Some of these metal components, particularly in the waste in SFL 5, also contain induced activity inside the metal. These radionuclides are not immediately accessible for dissolution in the water.

Solubility-limited dissolution

The dissolution of a substance that is in contact with water is limited by the solubility of the substance in the chemical environment in question. The chemical conditions that above all are of importance for solubility are pH, redox potential and the water's content of complexing agents. Solubilities for radionuclides and toxic metals at high pH and reducing conditions, which are the conditions expected to prevail inside waste packages in SFL 3-5, are summarized in Skagius *et al.* (1999a). The effect of the organic complexing agent isosaccharinic acid (ISA) on solubilities is also analyzed there. Owing to high pH and low concentration of carbonate ions inside the waste packages, complexation between metal ions and carbonate ions is negligible compared to the formation of metal hydroxide complexes (Skagius *et al.*, 1999a).

In the case of waste types that are stabilized in cement or concrete, sorption on the cement or concrete may increase the dissolution rate of the substance. The sorption reduces the concentration of the substance in the water, so that more of the substance in the solid phase can go into solution. However, the sorption also contributes to retaining the substance inside the waste package, see further Section 6.8.2.

Corrosion-limited dissolution

Activation products that have been formed in metal parts as a result of neutron irradiation are not immediately accessible for dissolution in the water in the waste packages. The same applies to the toxic metals in the waste, such as cadmium, beryllium and lead. Here it is the corrosion of the metal that determines the rate at which the radionuclides or the toxic metals dissolve, provided that the solubility of the substance under prevailing conditions is not exceeded.

The release of activation products in different types of steel and in Zircaloy in the waste in SFL 5 and some waste types in SFL 3 as the metal parts corrode has been calculated. These calculations are described in Skagius *et al.* (1999a).

Sorption and coprecipitation with corrosion products

The waste in SFL 3-5 consists largely of different kinds of steel. Corrosion of this steel gives rise to corrosion products, mainly iron oxides/hydroxides. Other metal ions that are present in small quantities in the water can sorb and be included in the corrosion products via coprecipitation. In order for the metal ion to form a true mixed phase with the corrosion product, it must fit into the corrosion product's crystal lattice. Metal ions that have chemical properties similar to those of iron could thus be coprecipitated with the corrosion products formed by steel corrosion. An example of such a metal ion is nickel. Corrosion of stainless steel and coprecipitation of the nickel present in the steel can reduce the solubility of nickel by a factor of 7 (Wiborgh, 1995).

Isotope dilution

The waste in SFL 3-5 contains both radioactive and stable isotopes of the same substance. This can affect the concentration of the radioactive isotope in the water if the substance in question has a limited solubility. The smaller the proportion of the radioactive isotope that is present in the waste, the lower will be its concentration in the water.

The waste in SFL 5 contains stainless steel and Zircaloy that have been exposed to neutron irradiation. Stainless steel contains nickel and Zircaloy contains zirconium. The proportion of ^{59}Ni in relation to the total quantity of nickel in different irradiated steel components in the waste has been estimated to be on average <1% by weight, and the proportion of ^{63}Ni <0.1% by weight (Skagius *et al.*, 1999a). The proportion of ^{93}Zr in relation to the total quantity of zirconium in Zircaloy components has been estimated to be 0.08 weight% (Skagius *et al.*, 1999a). This means that the solubility of ^{59}Ni and ^{63}Ni in waste packages with irradiated stainless steel components is about 100 and 1,000 times lower, respectively, than the total solubility of nickel. Similarly, the solubility of ^{93}Zr is about 1,000 times lower than the total solubility of zirconium.

6.8.2 Transport processes

Radionuclides and toxic metals that have dissolved in water inside waste packages can be transported out of the packages and further by advection and diffusion. Sorption retards transport through the barriers. Coprecipitation with concrete components can also contribute to retaining radionuclides and toxic metals. If the water contains colloids, they can sorb radionuclides and toxic metals and be transported with flowing water out to surrounding rock.

Advection

A prerequisite for advective outward transport of radionuclides and toxic metals is that there is some water flux in waste packages and surrounding barriers. This prerequisite is not met when the repository has been closed. However, gradients in water pressure and corrosion of steel containers, and eventually higher internal gas pressures, can increase the water permeability of the waste packages and concrete enclosures within a relatively short period of time, see Section 6.7. The gravel fill in the different repository parts has high water permeability.

Advective transport takes place in openings, gaps and cracks where the resistance to water flow is the least. In the gravel fill in the repository parts, the water flow is expected to be more evenly distributed in the space that is backfilled.

Gas that forms inside waste packages must expel some water in surrounding concrete in order to create transport pathways out. If this water contains radionuclides or toxic substances, the result may be elevated advective outward transport during a short period. Once gas channels have been created, the influence on the water flows and on advective transport in the barrier material is expected to be negligible, see Section 6.4.2.

Diffusion

Radionuclides and toxic pollutants can diffuse out through waste packages and surrounding barriers by following all water-filled openings, cracks and pore systems. The driving force for this diffusive transport is differences in concentration. The diffusive flow is also affected by the properties of the different barrier materials, such as porosity and the structure of pores and cracks.

Fresh, hydrated concrete has a complex pore system and low diffusivity. The effective diffusivity of tritiated water in a concrete made from Standard Portland Cement and sand has been experimentally determined to be $8 \cdot 10^{-12}$ m^2/s (Holgersson *et al.*, 1998a). Recrystallization of the CSH gel may with time change the pore structure to become less complex (Höglund and Bengtsson, 1991). This may lead to an increase in diffusivity in concrete. This internal transformation of the pore structure is, however, a slow process.

The composition of the water may also influence diffusion in porous materials. Anions have been shown in experiments to have lower diffusivity in clays and the micropores of the rock in water with a low ionic strength than in water with a high ionic strength. This phenomenon is termed anion exclusion and is explained by the fact that the pore surfaces that are negatively charged repel the anion. This reduces the porosity that is available for anion diffusion. At high ionic strengths, the negative charge of the pore surface decreases and the porosity available for diffusion increases (Olsson and Neretnieks, 1997; Yu and Neretnieks, 1997).

The water's ionic strength has also proved to be of importance for diffusion of cations that sorb via an ion exchange mechanism. At low ionic strengths, the diffusivity of these cations is higher than expected in clays and the micropores of the rock. One explanation for this is that the sorbed cations are mobile along the surface and that the driving force for this transport is the difference in the concentration of sorbed cations along the surface. At high ionic strengths, the sorption of the cations is lower and the driving force for the diffusion along the surface, the so-called surface diffusion, is thereby less.

Recently performed experiments have indicated the possibility that sorbed cations can repel other cations and thereby reduce their diffusivity in the pores of the rock (Holgersson *et al.*, 1998b). These experiments showed that the diffusion of Na^{2+} in pieces of rock was higher in a water with a lower Ca^{2+} concentration (equivalent to a non-saline concrete water) than in a water with a higher concentration of Ca^{2+} (equivalent to a concrete pore water after leaching-out of alkali hydroxides).

It is difficult to say anything about the possible importance of ionic strength and calcium concentration on diffusion in concrete, since diffusion measurements in concrete are as a rule performed on fresh concrete, which has a pore water with high ionic strength. However, diffusion measurements have been initiated in pieces of concrete that have been leached of all alkali hydroxide (Albinsson, personal communication, 1999), and the results of these tests may reveal the importance of ionic strength for diffusion in concrete.

Sorption

Radionuclides and toxic pollutants that dissolve in the waste can, via sorption, adhere to the barrier materials in the repository. This sorption may be reversible or irreversible, linear or non-linear, depending on the chemical form and concentration of the sorbing substance and on the mechanism by which sorption occurs.

The principal mechanisms for sorption of radionuclides on various minerals are considered to be ion exchange and surface complexation (Carbol and Engkvist, 1997; Bradbury and Baeyens, 1997). In ion exchange, an exchange takes place between cations, which neutralizes the negatively charged positions on the mineral surfaces and cations in the water. Sorption via ion exchange is dependent on the pH of the water, which affects the charge on the surfaces and the concentration of competing cations in the water. Sorption via surface complexation takes place by the formation of a so-called surface complex between functional groups on the mineral surface, usually hydroxyl groups, and molecules in the water (Dzombak and Morel, 1990). If the radionuclide or the toxic pollutant is easily hydrolyzed, chemical bonding takes place directly to the functional group on the surface. Surface complexation can also take place via electrostatic attraction between the charged surface groups and the substances in the water. The degree of surface complexation is dependent on both the ionic strength and the pH of the water. The ionic strength affects the charge of the surfaces and the pH affects both the charge of the surfaces and how easily the dissolved substances are hydrolyzed.

Normally the sorption is described in simplified form as a distribution coefficient, K_d, between the substance sorbed on the surface and dissolved in the water. This coefficient is not a general constant, but is dependent on the prevailing conditions, for example the water's pH, ionic strength and content of strong complexing agents. The composition of the solid phase is also of importance. K_d values should therefore be determined under conditions that are as similar as possible to those that will prevail in the repository. K_d values for radionuclides and toxic metals on concrete and gravel fill at the pHs and ionic strengths that will prevail in SLF 3-5 are summarized in Skagius *et al.* (1999a).

Isosaccharinic acid, ISA, formed by the alkaline hydrolysis of cellulose, can form complexes with radionuclides and toxic metals and thereby influence their sorption. Organic materials that may contain cellulose – for example wood, paper, cloth – are present in the waste in SFL 3. Experimental investigations of how ISA affects sorption on cement show that concentrations of ISA in the water that are greater than 10^{-4} M may also reduce the sorption of radionuclides (Bradbury and Van Loon, 1998). This effect fades with time on a timescale of months to years (Holgersson *et al.*, 1998a). Furthermore, ISA itself sorbs on cement, which contributes to maintaining low concentrations of ISA in the water (Bradbury and Van Loon, 1998; Holgersson *et al.*, 1998a).

Precipitation/coprecipitation with calcite

Calcite may precipitate in the concrete and the gravel backfill in SFL 3-5. This may reduce the concentration of inorganic ^{14}C present as carbonate in the aqueous phase. Other divalent cations in the water with a smaller ionic radius than calcium (manganese, zinc, iron, cadmium, cobalt and magnesium) may coprecipitate with calcite by replacing calcium in the crystal lattice (Meece and Benninger, 1993).

Transport with colloids

Radionuclides and toxic metals that sorb readily on minerals may also adhere to colloidal particles in the water. These particles may then be transported by flowing water and by diffusion. The colloids may be inorganic mineral particles or organic particles, e.g. bacteria and humus. Owing to high calcium concentrations, the amount of colloids in the repository is expected to be small, see Sections 6.6.1 and 6.6.2. Furthermore, colloidal particles are probably negatively charged. Transport of such particles in pores and cracks in concrete is therefore counteracted by the negatively charged pore surfaces (Wiborgh, 1995).

Small quantities of colloids in the water mean that the capacity of the colloids to sorb and transport radionuclides is small compared with the rock's capacity to sorb and retard transport. This has already been shown in an evaluation of the effect of transport with colloids from a repository with spent fuel (Allard *et al.*, 1991).

6.8.3 Handling in the safety assessment

In the safety assessment, all radionuclides, except for activation products in steel and Zircaloy, have been assumed to be available for immediate dissolution in water in waste packages. The concentration of the radionuclide in the water is determined by its quantity in the waste and sorption on cement and/or concrete in the waste package, or its solubility. The effect of isotope dilution on the solubility of nickel isotopes and zirconium has not been taken into account, however. We assume that the activation products go into solution as the metals corrode. Adsorption or coprecipitation with corrosion products is neglected, however.

The toxic metals lead, beryllium and cadmium that are included in the safety assessment are assumed to dissolve immediately in the water, regardless of the form in which they are present in the waste. Their concentration is determined by their quantity in the waste and sorption on cement and/or concrete in the waste package, or the solubility of the metal.

Sorption and solubility data have been chosen based on the hydrochemical conditions described in Section 6.6.3. Some waste types in SFL 3 contain organic materials with cellulose. Solubilities both without and with the complexing agent ISA have therefore been used. Sorption is assumed to be unaffected by ISA, however, since the expected ISA concentrations in waste packages with cellulose are less than 10^{-4} M (Skagius *et al.*, 1999a).

Radionuclides and toxic pollutants dissolved in the water in the waste packages are assumed to be transported out through the waste packages and surrounding barriers by advection and diffusion. Sorption on concrete and gravel backfill retards the transport of sorbing substances. Steel containers surrounding the waste are completely disregarded, in terms of both resistance to advective and diffusive transport and possible sorption and coprecipitation with corrosion products. Retardation of inorganic ^{14}C has been taken into account as a sorption process in the materials, despite the fact that it is more a question of a precipitation in the form of calcite. It is assumed that coprecipitation of divalent cations with calcite in the barriers does not take place.

Diffusivities and sorption data are selected based on the barrier properties and hydrochemical conditions described in Sections 6.7.3 and 6.6.3. Water flows in the different repository parts and the distribution of the water flow within each repository part are based on the results of the generic hydrological near-field model.

Transport of radionuclides and toxic pollutants with colloids has not been included in the safety assessment, since the quantity of colloids is expected to be very small.

The assumptions that underlie the reference case in the safety assessment are described in greater detail in Chapter 8.

6.9 Release of radioactive gas

The waste in SFL 3-5 does not contain any radionuclides in the gaseous phase. Of the radionuclides present in the waste, only organic ^{14}C is expected to be able to be emitted and transported out of the near field as a gas, for example if ^{14}C forms methane and mixes with gas from metal corrosion or degradation of organic materials. This outward transport of gas can take place as soon as transport pathways for gas have been created in the near-field barriers.

The safety assessment includes a simple estimation of release of organic ^{14}C as a gas. This is described in greater detail in Chapter 8.

6.10 References

Albinsson Y, 1999. Department of Nuclear Chemistry, Chalmers University of Technology, Gothenburg.

Allard B, Borén H, Ephraim J, 1995. Cellulosa i cementsystem. Nedbrytningsprodukter och deras effekter på radionukliders transportegenskaper. SKB rapport SFR 95-01, Svensk Kärnbränslehantering AB, Stockholm.

Allard B, Karlsson F, Neretnieks I, 1991. Concentrations of particulate matter and humic substances in deep groundwaters and estimated effects on the sorption and transport of radionuclides. SKB Technical Report TR 91-50, Svensk Kärnbränslehantering AB, Stockholm.

Andersson K, Allard B, 1986. The chemical conditions within a cement-containing radioactive waste repository. SKB report SFR 86-08, Svensk Kärnbränslehantering AB, Stockholm.

Bateman K, Coombs P, Noy D J, Pierce J M, Wetton P, 1995. Nagra/Nirex/SKB column experiments: Results of experiments and modelling. BGS Report WE95/25C and SKB report AR 95-16, Svensk Kärnbränslehantering AB, Stockholm.

Bradbury M H, Baeyens B, 1997. Far-field sorption data bases for performance assessment of a L/ILW repository in an undisturbed palfris marl host rock. PSI Bericht Nr. 97-16, Paul Scherrer Institut, Villingen, Schweiz.

Bradbury M H, Van Loon L R, 1998. Cementitious Near-Field Sorption Data Bases for Performance Assessment of a L/ILW Repository in a Palfris Marl Host Rock. CEM-94: UPDATE I, June 1997. PSI Bericht Nr. 98-01, Paul Scherrer Institut, Villingen, Schweiz.

Bradshaw S, Gaudie S C, Greenfield B F, Lyon C E, Rees J H, Spindler M W, Wilkins J D, 1986. Preliminary experimental studies on the chemical and radiation degradation of combustionable plutonium contaminated material. AERE-R12223, AEA Technology, Harwell, UK.

Bradshaw S, Gaudie S C, Greenfield B F, Long S, Rees J H, Spindler M W, Wilkins J D, 1987. Experimental studies on the chemical and radiation decomposition of intermediate-level wastes containing organic materials. AERE-R12806, AEA Technology, Harwell, UK.

Brown D A, Sheriff B L, 1998. The effects of microbial activity on the natural and engineered barriers of a high-level nuclear waste repository. SKI report 98:24, Swedish Nuclear Power Inspectorate, Stockholm.

Carbol P, Engkvist I, 1997. Compilation of radionuclide sorption coefficients for performance assessment. SKB report R-97-13, Svensk Kärnbränslehantering AB, Stockholm.

Dzombak D A, Morel M M, 1990. Surface complexation modelling. Hydrous ferric oxide. John Wiley & Sons, New York.

Engkvist I, Albinsson Y, Johansson-Engkvist W, 1996. The long-term stability of cement - Leaching tests. SKB Technical Report TR 96-09, Svensk Kärnbränslehantering AB, Stockholm.

Greenfield B F, Moreton A D, Spindler M W, Williams S J, Woodwark D R, 1992. The effects of the degradation of organic materials in the near field of a radioactive waste repository. In Scientific Basis for Nuclear Waste Management XV, pp 299-306.

Greenfield B F, Holtom G J, Hurdus M H, O´Kelly N, Pilkington N J, Rosevear A, Spindler M W, Williams S J, 1995. The identification and degradation of isosaccharinic acid, a cellulose degradation product. Mat. Res. Soc. Symp. Proc**.** 353, p 1151.

Holgersson S, Albinsson Y, Allard B, Borén H, Pavasars I, Engkvist I, 1998a. The effects of Gluco-isosaccharinic acid on Cs, Ni, Pm and Th sorption onto, and diffusion into cement. Radiochim. Acta, 82, pp 393-398.

Holgersson S, Albinsson Y, Engkvist I, Rochelle C, Pierce J, 1998b. Interactions of cement pore fluids with host rock and the effects on HTO, Na and Cs diffusion. Radiochem. Acta, 82, pp 197-203.

Holmén J, 1997. On the flow of groundwater in closed tunnels. Generic hydrogeological modelling of nuclear waste repository, SFL 3-5. SKB Technical Report TR 97-10, Svensk Kärnbränslehantering AB, Stockholm.

Hunter R J, 1987. Foundations of colloid science. Volume I. Oxford Science Publications, Clarendon Press, Oxford, Chapter 2, pp 93-95.

Höglund L O, Bengtsson A, 1991. Some chemical and physical processes related to the long-term performance of the SFR repository. SKB report SFR 91-06, Svensk Kärnbränslehantering AB, Stockholm.

Höglund L O, 1992. Some notes on ettringite formation in cementitious materials; influence of hydration and thermodynamic constraints for durability. Cement and Concrete Research, 22, pp 217-228.

Höglund L O, 1993. Long-term function of concrete grout in SFR. SKB report SFR 93-01, Svensk Kärnbränslehantering AB, Stockholm.

Karlsson F, Lindgren M, Skagius K, Wiborgh M, Engkvist I, 1999. Evolution of the geochemical conditions in SFL 3-5. SKB report R-99-15, Svensk Kärnbränslehantering AB, Stockholm.

Laaksoharju M, Degueldre C, Skårman Ch, 1995. Studies of colloids and their importance for repository performance assessmet. SKB Technical Report TR 95-24, Svensk Kärnbränslehantering AB, Stockholm.

Lessart P, Libert M F, Sellier R, Bilcot J B, Besnainou B, Camaro S, Bernat P, Perfettini J, Grec D, Rouquette F, Rosevear A, O´Kelly N, 1996. Container property ensuring safety: gas emission, biodegradation, corrosion. European Commission Report EUR 17103 EN.

Lindgren M, Pers K, Skagius K, Wiborgh M, Brodén K, Carlsson J, Riggare P, Skogsberg M, 1998. Low and intermediate level waste in SFL 3-5: Reference inventory. A compilation of low and intermediate level waste to be sent to SFL 3-5. Reg. No: 19.41/DL31. SKB, Svensk Kärnbränslehantering AB, Stockholm.

Meece D E, Benninger L K, 1993. The coprecipitation of Pu and other radionuclides with $CaCO_3$. Geochim. et Cosmochim. Acta, **57**, pp 1147-1458.

Moreno L, Neretnieks I, 1991. Some calculations of radionuclide release from the silo repository. SKB rapport SFR 91-07, Svensk Kärnbränslehantering AB, Stockholm.

Ohlsson Y, Neretnieks I, 1997. Diffusion data in granite. SKB Technical Report TR 97-20, Svensk Kärnbränslehantering AB, Stockholm.

Pavasars I, 1999. Characterisation of organic substances in waste materials under alkaline conditions. Doctoral thesis, Tema Vatten, Linköping University, Linköping Studies in Art and Science, No 196, Linköping.

Savage D, Bateman K, Hill P, Hughes C, Milodowski A, Pearce J, Rae E, Rochelle C, 1992. Rate and mechanism of the reaction of silicates with cement pore fluids. Applied Clay Science, **7**, pp 33-45.

Savage D, 1997. Review of the potential effects of alkaline plume migration from a cementitious repository for radioactive waste. Implications for performance assessment. R&D Technical Report P60, Environment Agency, Bristol.

Skagius K, Pettersson M, Wiborgh M, Albinsson Y, Holgersson S, 1999a. Compilation of data for analysis of radionuclide migration from SFL 3-5. SKB report R-99-13, Svensk Kärnbränslehantering AB, Stockholm.

Skagius K, Lindgren M, Pers K, 1999b. Gas generation in SFL 3-5 and effects on radionuclide release. SKB report R-99-16, Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1993. Slutförvar för radioaktivt driftavfall – SFR 1. Slutlig Säkerhetsrapport, reviderad utgåva – maj 1993. Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1999. SR 97 - Main Report. Svensk Kärnbränslehantering AB, Stockholm.

Smellie J (ed.), 1999. Maqarin natural analogue study. Phase III. SKB Technical Report TR 98-04, Volume I and II. Svensk Kärnbränslehantering AB, Stockholm.

Thunvik R., Braester C, 1986. Calculation of gas migration in fractured rock. SKB report SFR 86-04, Svensk Kärnbränslehantering AB, Stockholm.

Wiborgh M, 1995. Prestudy of final disposal of long-lived low and intermediate level waste. SKB Technical Report TR 95-03, Svensk Kärnbränslehantering AB, Stockholm.

Wikramaratna R S, Goodfield M, Rodwell W R, Nash P.J., Agg P J, 1993. A preliminary assessment of gas migration from the copper/steel canister. SKB Technical Report TR 93-31, Svensk Kärnbränslehantering AB, Stockholm.

Yu Ji-Wei, Neretnieks I, 1997. Diffusion and sorption properties of radionuclides in compacted bentonite. SKB Technical Report TR 97-12, Svensk Kärnbränslehantering AB, Stockholm.

7 Hydrology calculations

7.1 Introduction

The preliminary safety assessment of SFL 3-5 is primarily focused on the performance of the near-field barriers. The water flow through the different barriers in the near field has been calculated with a generic hydrological model of the near field. The calculations have been carried out to investigate the importance of the hydraulic conductivity of the barriers, and above all what the contrasts in conductivity entail. The barriers in the near field comprise the rock around the repository and the various engineered barriers, such as gravel backfill, concrete walls, porous concrete, packages, plugs, etc. (Holmén, 1997).

So far the calculations are completely general and not associated with any special site. But in order to obtain the flow in the near field when the repository is situated at Aberg, Beberg and Ceberg, data are needed on the water flow in the rock on these sites. The ambition has been to choose input data that agree with those used for the deep repository for spent fuel at Aberg, Beberg and Ceberg. The agreement cannot be complete, since SFL 3-5 lies on the outskirts of or even outside the study areas for the deep repository for spent fuel. Nor is SFL 3-5 included in the local hydrology models, which describe in detail the water flow on the different sites, but this has not been necessary since the goal of the assessment of SFL 3-5 is more limited. Specific water flows for SFL 3-5 have instead been calculated with the aid of data from the general regional hydrology models developed for the three sites.

Water flows and flow directions in the rock are also needed to calculate transport of radionuclides with the groundwater in the far field, i.e. between the repository and the biosphere. We have used data from the regional hydrology models for this purpose as well.

This chapter begins in Section 7.2 by examining some general aspects of the groundwater flow in tunnels and rock caverns. The calculations of the water flow in the near-field barriers performed with the aid of the generic hydrological model are then summarized in Section 7.3. The calculations made of the direction and size of the groundwater flow in the rock on the three sites Aberg, Beberg and Ceberg with the proposed positioning of SFL 3-5 are summarized in Section 7.4. The same section also presents estimated flow paths and advective travel times for water from SFL 3-5 to the ground surface on the different sites. In conclusion, Section 7.5 provides a compilation of the hydrological data chosen as input data to the migration calculations.

7.2 General about groundwater flow in tunnels and rock caverns

7.2.1 Hydraulic life cycle of the repository

The repository consists of rock caverns and tunnels. The facility's life cycle has four phases with different hydraulic conditions: construction, operation, closure and post-closure.

- *Construction.* The facility's tunnels and rock caverns are built and groundwater flows into them. The tunnels and rock caverns are drained of inflowing groundwater, which means that the water pressures in the surrounding rock mass will be lower than the natural pressures that existed before the facility was built.
- *Operation.* The facility is kept open, ventilated and drained while waste is deposited.
- *Closure.* Tunnels and rock caverns are backfilled and sealed. Inflowing groundwater fills the facility when it is no longer drained. The groundwater pressures in tunnels, rock caverns and surrounding rock mass will rise during this period until a steady state has been achieved.
- *Post-closure phase.* This is the very long period of time during which the activity in the waste decays to an acceptably low level. Tunnels, rock caverns and ventilation shafts are backfilled. The groundwater system around the facility is in a steady state, large pressure changes take place only slowly.

7.2.2 Groundwater flow in rock and tunnels

The natural groundwater flow in a rock mass is controlled by its hydraulic properties, the local and regional topography, and by groundwater recharge. The repository facility will be located in a crystalline rock. The groundwater flow in such a rock mass goes mainly in fractures and fracture zones of differing size and direction. The direction and size of the groundwater flow may therefore deviate substantially from the mean values obtained for a larger regional scale.

A tunnel receives groundwater along an inflow section and delivers groundwater along an outflow section. The size of the flow varies along the tunnel. The inflow section (upstream) and downflow section (downstream) are dependent on the direction of the tunnel and the direction of the regional groundwater flow.

The size of the groundwater flow in a tunnel can be specified as "specific flow" and "total flow". Specific flow is defined as flow per unit area ($m^3/(m^2s) = m/s$) and provides information on the flow at a given point. The arithmetic mean of the specific flow in the tunnel is used in the calculations of water flows in the near field with the generic model (Holmén, 1997). Total flow is defined as the quantity of water that flows into a tunnel (m^3/s). In a steady state, the quantity of water that flows out is equally great, provided the tunnel is not drained. The calculation of the total flow is based on the mass balance taken over the envelope surface area of the studied tunnel. In tunnel systems, water can flow into and out of the tunnel system at several places. The total flow provides information on how much water “visits” a tunnel.

7.2.3 Groundwater flow in tunnels during the post-closure phase

In a backfilled and closed tunnel where the groundwater system around the facility is in a steady state, the flow is dependent on: (i) the shape, length and cross-sectional area of the tunnel, (ii) the hydraulic properties (conductivity) of the backfill in the tunnel, (iii) the size and direction of the regional groundwater flow, and (iv) the heterogeneity of the surrounding rock mass.

The size of the flow in a highly conductive tunnel is mainly dependent on the conductivity of the surrounding rock mass. But the direction of the regional flow is also of importance. The specific flow is greatest if the regional flow is directed along the tunnel. The flow is smallest if the regional flow is directed perpendicular to the tunnel. The same applies to the total flow in the tunnel.

In the case of a tunnel that is more conductive than the surrounding rock mass, the groundwater flow will converge on the tunnel’s inflow section. In the inflow section of the tunnel, the flow inside the tunnel will increase from the inflow section furthest away towards the middle of the tunnel. The maximum flow occurs in the middle of the tunnel (between the inflow section and the outflow section). In the outflow section of the tunnel, the flow inside the tunnel will decrease from the middle of the tunnel towards the outflow section situated furthest away. From the outflow section of the tunnel, the groundwater will flow from the tunnel towards the surrounding rock mass.

In the calculations we assume that the rock mass is homogeneous. This can lead to underestimation of the flow in the tunnel. The difference in calculated flows between a homogeneous and a heterogeneous rock mass become negligible when the tunnel is sufficiently long. In the case of a rock mass with a heterogeneity of the rock in the Äspö hard rock laboratory, HRL, the difference becomes negligible when the tunnel is approximately 700 m long and the regional flow is directed along the tunnel (Holmén, 1997). If the regional flow is perpendicular to the tunnel, the difference becomes negligible at a tunnel length of about 2,000 m. Note that this applies to the total flow in the tunnel. In the case of a 100 m long tunnel with a cross-sectional area of 100 m^2 positioned in a heterogeneous rock mass like the one at the Äspö HRL, the expected total flow in the tunnel is 2 to 3 times greater than the calculated total flow with the assumption of homogeneity.

In SFL 3 and SFL 5, the waste is emplaced inside a concrete enclosure (encapsulation). The backfill outside the concrete enclosure consists of crushed rock, which is more conductive than the surrounding rock mass. Groundwater from the surrounding rock mass therefore flows mainly through the backfill, around and outside the encapsulation. In this way the groundwater flow through the encapsulation is reduced by several orders of magnitude. The contrasts in conductivity are crucial to the distribution of the flow. The greater the contrast in conductivity between the flow barrier (backfill) and the encapsulation on the one hand and the surrounding rock mass on the other, the less the flow through the encapsulation.

In summary, we observe that the flow through a tunnel is dependent on the magnitude of the regional groundwater flow. This is in turn influenced by topography, groundwater recharge and the conductivity of the rock mass. The following conclusions apply regardless of the regional conditions, however:

- If tunnel conductivity is great and the tunnel is long, the most important parameter is the direction of the regional flow.
- If a tunnel is shorter than about 250 m, heterogeneity is probably the most important parameter.
- The flow through an encapsulation surrounded by a backfill is mainly dependent on the conductivity of the backfill.

7.3 Groundwater movements in the near field

7.3.1 Hydrological model of the near field

The hydrological model of the near field is based on a detailed description of the different parts of the repository and the rock mass around the repository (Holmén, 1997). All of SFL 3-5 is included, with the exception of any elevator shafts and connecting tunnels. We assume that all of these are sealed in such a manner that we can disregard their influence. Plugs have been emplaced where SFL 3 and SFL 5 connect to SFL 4, see Chapter 3. The calculation model is three-dimensional and is based on a continuum description that was drawn up with the aid of a method using finite differences (GEOAN). All calculations that have been performed pertain to the post-closure phase, when both water flows and pressure differences have reached a steady state.

A horizontal cross section through the repository model is shown in Figure 7-1. The waste packages, the concrete enclosure and the backfilled porous concrete in the concrete enclosure in SFL 3 and SFL 5 comprise a homogeneous unit (the encapsulation) with a conductivity that is assumed to be 10 times higher than in the rock mass. The backfill outside the concrete enclosure in SFL 3 and SFL 5 comprises a homogeneous flow barrier. In the calculations, the conductivity in the flow barrier is varied to simulate a) a backfill of crushed rock that is more conductive than the surrounding rock, and b) a backfill that is less conductive than the surrounding rock, e.g. sand/bentonite or bentonite. SFL 4 is described in the model as a tunnel that is either empty or filled with sand or gravel.

The plugs consist of both concrete and bentonite, where the concrete is assumed to have a conductivity equal to that of the rock mass and the bentonite a conductivity 10 times lower than that of the rock mass. In the model, these plugs are positioned where SFL 3 and SFL 5 connect to SFL 4, see Figure 7-1.

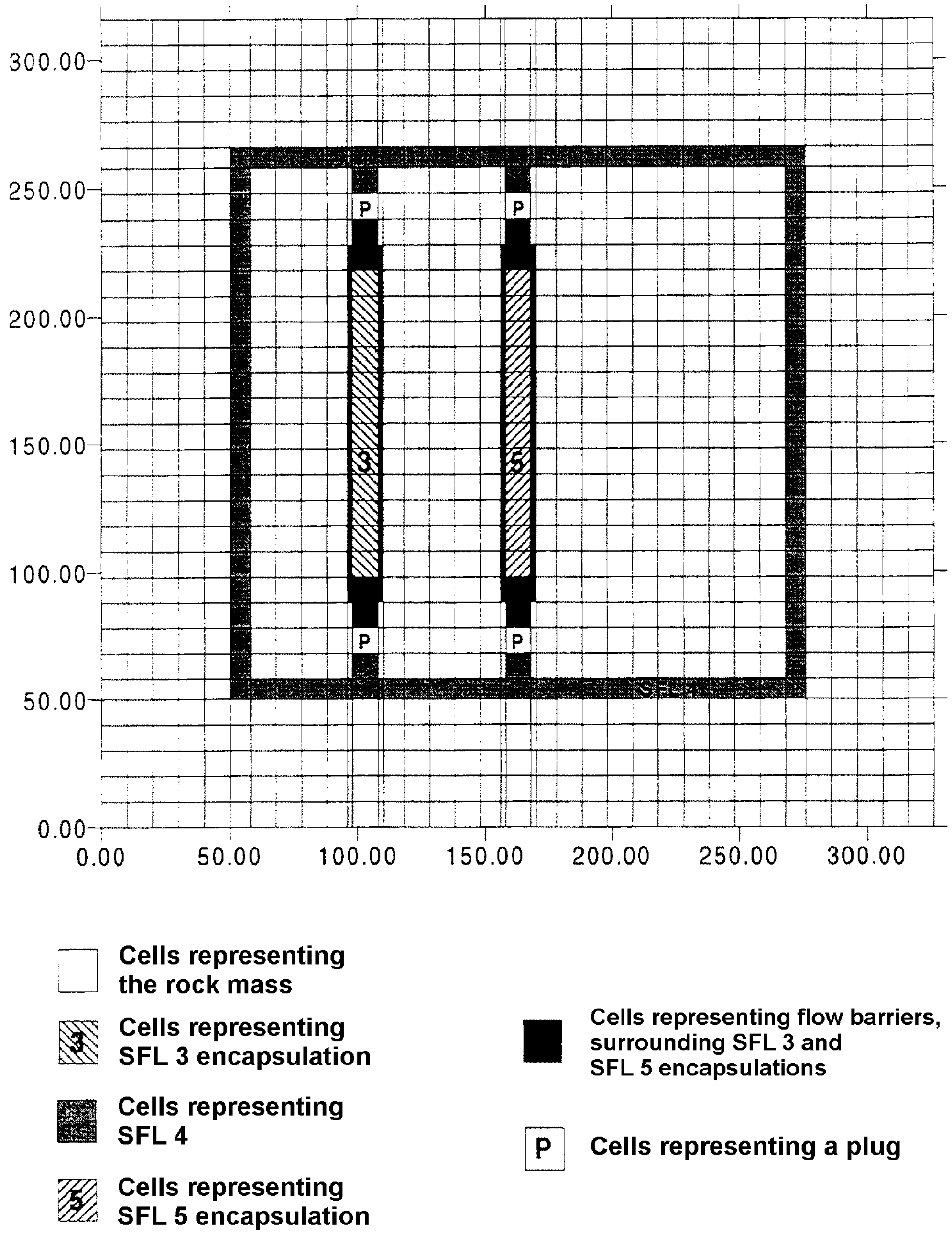

Figure 7-1 Horizontal cross-section through the model used to calculate water flows in the barriers in SFL 3-5, scale in metres. (Holmén, 1997)

7.3.2 Importance of direction of regional groundwater flow

The calculations show that the total flow in the flow barriers for SFL 3 and SFL 5 varies by less than a factor of 2 with the direction of the regional flow. The largest flow in the flow barrier is obtained for either a vertical regional flow or for a horizontal regional flow, directed along SFL 3 and SFL 5.

The total water flow through the encapsulations in SFL 3 and SFL 5 varies by less than a factor of 3 with the direction of the regional flow.

Depending on the direction of the regional flow, the total flow in the SFL 4 tunnel varies by less than a factor of 5. The largest flow in the tunnel is obtained for a horizontal regional groundwater flow, directed perpendicular to SFL 3 and SFL 5.

7.3.3 Importance of conductivity in backfill in SFL 3 and 5

The importance of the conductivity in the flow barriers in SFL 3 and SFL 5 was studied for widely differing values of the conductivity, from very low to very high conductivity relative to the rock mass (by a factor of 0.001–100,000). The purpose was to see what the effects of a low-conductive versus a permeable backfill would be. The conductivity in the backfill in SFL 4 was assumed to be 10,000 times higher than in the rock mass. The calculations were performed for three different assumptions regarding the direction of the regional flow, i.e. horizontal flow along SFL 3 and SFL 5, horizontal flow perpendicular to the rock vaults, and finally vertical flow.

The results of the calculations show that the water flow in SFL 3’s and SFL 5’s flow barriers (backfill) will increase with increasing conductivity, up to a given value when the flow is instead determined by the conductivity of surrounding rock. The calculations show that the specific flow in the flow barriers does not reach higher than about 25 times the regional flow, and that this occurs when the conductivity is of the order of 10,000 times higher than in surrounding rock.

The flow in SFL 3’s and SFL 5’s encapsulations is reduced by flow barriers with high conductivity. If, for example, the encapsulation’s barriers have 100,000 times higher conductivity than the rock mass, the specific flow through the repository will be at most about 0.003 times the regional flow. Flow barriers with low conductivity can naturally also reduce the flow through the encapsulation, but to bring about a significant reduction the conductivity must be about 10 times lower than in the rock mass.

The flow in SFL 4 is affected only very slightly by the conductivity in the flow barriers in SFL 3 and SFL 5.

7.3.4 Importance of conductivity in backfill in SFL 4

The importance of the conductivity in the backfill in SFL 4 was also studied for different conductivity values (0.1–100,000 times the rock mass). The calculations were performed for the same three principal directions of the regional flow as in the calculations for SFL 3 and SFL 5 (see above).

The results of the calculations show that the specific flow in the SFL 4 tunnel will be at most about 300 times higher than the regional flow. This maximum flow is obtained when the contrast in conductivity between the tunnel and surrounding rock is 10,000 times or more.

The long SFL 4 tunnel that surrounds SFL 3 and SFL 5 can act as a hydraulic cage. However, this presupposes that the regional flow is horizontal or nearly horizontal and that the conductivity in the SFL 4 tunnel is at least 100 times higher than in the rock mass. The greater the conductivity in SFL 4, the less the flow in SFL 3 and SFL 5. But the flow in SFL 3 and SFL 5 is not affected if the conductivity in the SFL 4 tunnel is less than 100 times that of the rock mass, or if the direction of the regional flow is vertical or nearly vertical.

7.3.5 Importance of plugs in the tunnels

The importance of plugs in the repository for the water flows in SFL 3-5 was studied by performing calculations with and without plugs. Four plugs were positioned where SFL 3 and SFL 5 connect to SFL 4, see Figure 7-1. Without these four plugs, the total flow in SFL 4 will be at most about twice as high compared with if the plugs are in place. The total flow in SFL 3 and SFL 5 will also be higher, at most about 6 times in the encapsulations and about 4 times in the flow barriers.

We also tried including 4 extra plugs in the SFL 4 tunnel, but the effect of that measure on the flow in e.g. SFL 4 is insignificant.

7.3.6 Importance of the heterogeneity of the rock mass

All results reported above were obtained under the assumption that the rock mass is homogeneous. This is a simplification that greatly facilitates the calculations. To estimate the importance of the heterogeneity of the rock mass, results were used from calculations with a stochastic continuum model where the rock mass has properties similar to those at the Äspö HRL. The results show that in such a rock mass, the heterogeneity leads to an increase of the total flow in SFL 4 by a factor of about 2. In SFL 3 and 5, the maximum flow in a flow barrier will increase by a factor of about 2.5 times, provided that the conductivity of the barrier is higher than that of the rock mass. The aforementioned flows pertain to the expected total flows, i.e. the most probable ones.

7.4 Groundwater movements in the far field

7.4.1 Models and methods

Regional hydrology models have been used to calculate the size, directions and advective travel times for the water flow in the far field to SFL 3-5 at the sites Aberg, Beberg and Ceberg. The models were originally set up for studies of the deep repository for spent fuel, but they have also been able to be used here since they cover the areas where SFL 3-5 have been positioned (see Chapter 4) and are sufficiently accurate for our preliminary assessment.

The model used for Aberg covers a square area of about 100 km^2 and a depth of 3 km. The rock is regarded in the model as a stochastic continuum. Boundary conditions and other input data to the calculations are the ones given as natural conditions in the development and calibration of the model (Svensson, 1997a).

The model for Beberg covers an area of about 75 km^2 and a depth of about 2 km. The model for Ceberg covers an area of about 300 km^2 and a depth of 1.5 km. The rock is regarded as a porous medium in both models. The input data and boundary conditions used in the "base case" in the calculations for Beberg are the ones that gave the best agreement with observed conditions at Beberg in the calibration of the model (Hartley *et al.*, 1998). The large regional model with a head boundary condition on the surface given by the topography was used for the "base case" at Ceberg (Boghammar *et al.*, 1997).

The flow paths from the SFL 3-5 repository to the surface have been studied by releasing particles in those parts of the regional models that correspond to the repository area (Svensson, 1997b; Hartley and Lindgren, 1997). 10,000 particles were released in the model for Aberg (of which nine have been studied in greater detail), while 36 particles were released at Beberg and Ceberg. The water's advective travel times were calculated, along with the length of flow paths from the repository to the surface and flow rates and directions in relation to the horizontal plane of the water flow in the elements in the model that correspond to the proposed positioning of SFL 3-5.

The advective travel time is a theoretical quantity used in the transfer of results from calculations with hydro models to transport models. In the later case advective travel times are used to calculate the time for transport of dissolved substances from repository level up to the surface. The transport times are generally several orders of magnitude higher than the advective travel times.

7.4.2 Direction and size of groundwater flow

The calculated water flows and the direction of the flow in relation to the horizontal plane in the elements that correspond to the positioning of SFL 3-5 are given in Table 7-1. The mean value of the specific water flow varies between 0.04 and 27 l/m^{2}y, the lowest value being for Ceberg and the highest for Aberg. The direction of the mean flow is roughly horizontal at Beberg and Ceberg, while it is directed about 30 degrees downward at Aberg.

Table 7-1 Specific water flows and flow directions at the positions for SFL 3-5 at Aberg, Beberg and Ceberg.

	Specific water flow (l/m^2/y)[a)]			**Angle of flow to horizontal**		
	Aberg	**Beberg**	**Ceberg**	**Aberg**	**Beberg**	**Ceberg**
Mean	27	3	0.04	Down, 31°	Down, 6°	Down, 1°
Min.	6	2	0.04	Down, 84°	Down, 9°	Down, 18°
Max.	64	4	0.05	Up, 11°	Down, 2°	Up, 17°

a) Darcy velocity

7.4.3 Flow paths and advective travel times for ground water

The calculated lengths of flow paths and advective travel times for water from the repository to the surface are summarized in Table 7-2. The results are based on nine of the particles released in the model for Aberg and on all 36 particles released in the models for Beberg and Ceberg (Svensson, 1997b; Hartley and Lindgren, 1997).

The proposed positioning of SFL 3-5 at Aberg puts the repository in a fracture zone (Svensson, 1997b), which means short flow path lengths and short advective water travel times. The mean travel time is calculated to be 13 years and the mean transport length for the particles is 505 m. It should, however, be pointed out that the simulation time in the calculation was limited to 25 years and that two of the nine particles were then still 100 m below the ground surface. Longer travel times and transport lengths were obtained for Beberg and Ceberg, with means of 40 years and 1,125 m at Beberg and 906 years and 1,396 m at Ceberg (Hartley and Lindgren, 1997).

Table 7-2 Advective travel times and transport lengths for particles released at the positions for SFL 3-5 at the three sites Aberg (nine particles), Beberg and Ceberg (36 particles).

	Transport length (m)			**Advective travel time (yrs)**		
	Aberg	**Beberg**	**Ceberg**	**Aberg**	**Beberg**	**Ceberg**
Mean	505	1,125	1,396	13	40	906
Std.dev.	132	194	93	7	24	208
Min.	361	757	1,238	7	7	653
Max.	728	1,479	1,583	25 [a)]	80	1,292

[a)] Simulation time 25 years – two of the nine particles still 100 m below the ground surface.

Figure 7-2 shows the start and end positions for 1,000 of the 10,000 particles that were released in the model for Aberg. The end position for most of the particles is in the water east and north of Äverö, about 300 m east of and 600 m north of SFL 3-5.

Start and end positions for the 36 particles released in the model for Beberg are shown in Figure 7-3, while Figure 7-4 shows start and end positions for the 36 particles released in the model for Ceberg. At Beberg, all particles came up in a fracture zone, Imundbo, and at Ceberg they came up in roughly the same place near the Husån river.

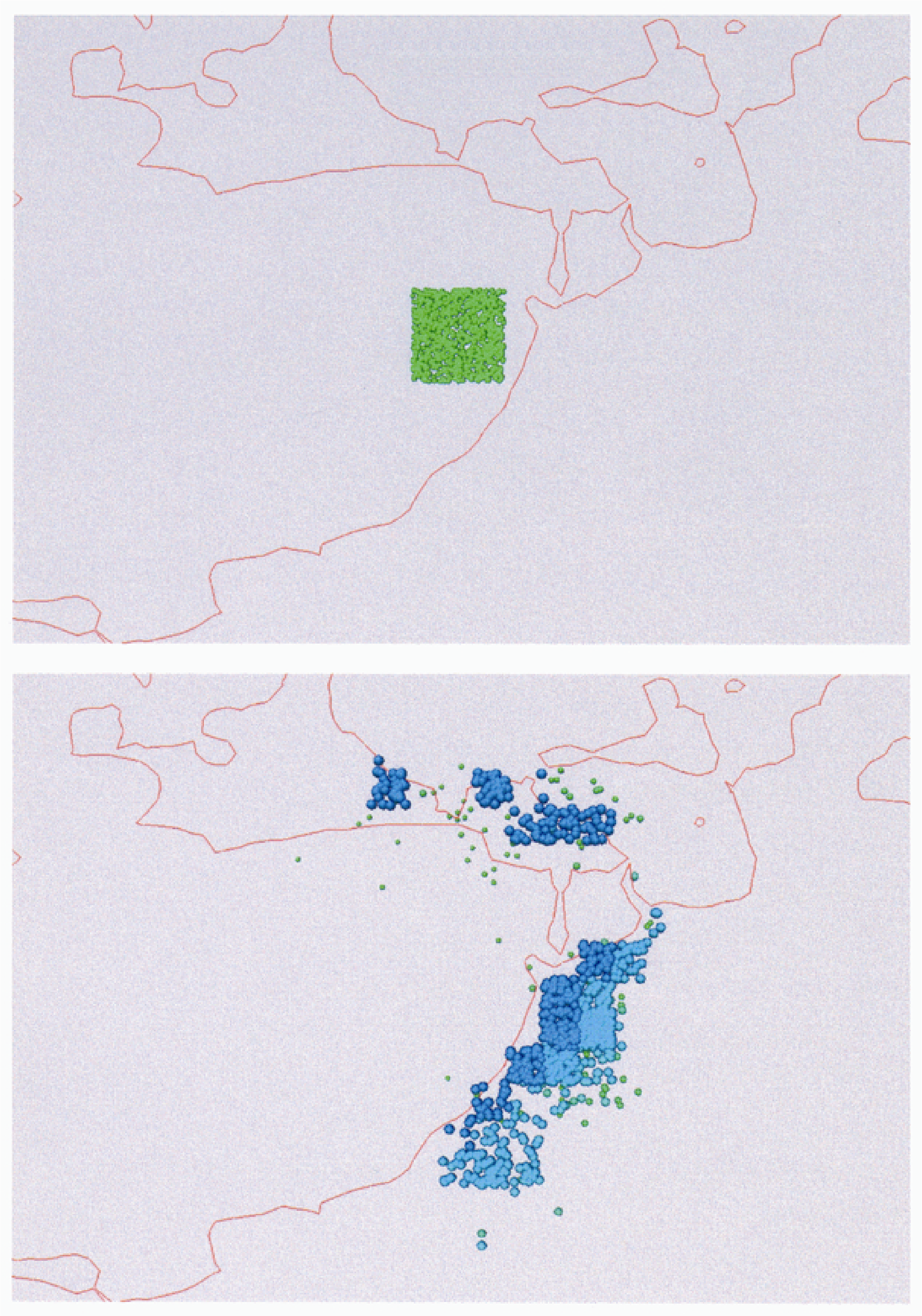

Figure 7-2 Start positions (upper figure) and position after 25 years (lower figure) for 1,000 particles at Aberg. Large blue marks indicate that the particles are at the surface, while smaller blue marks indicate that the particles are deeper.

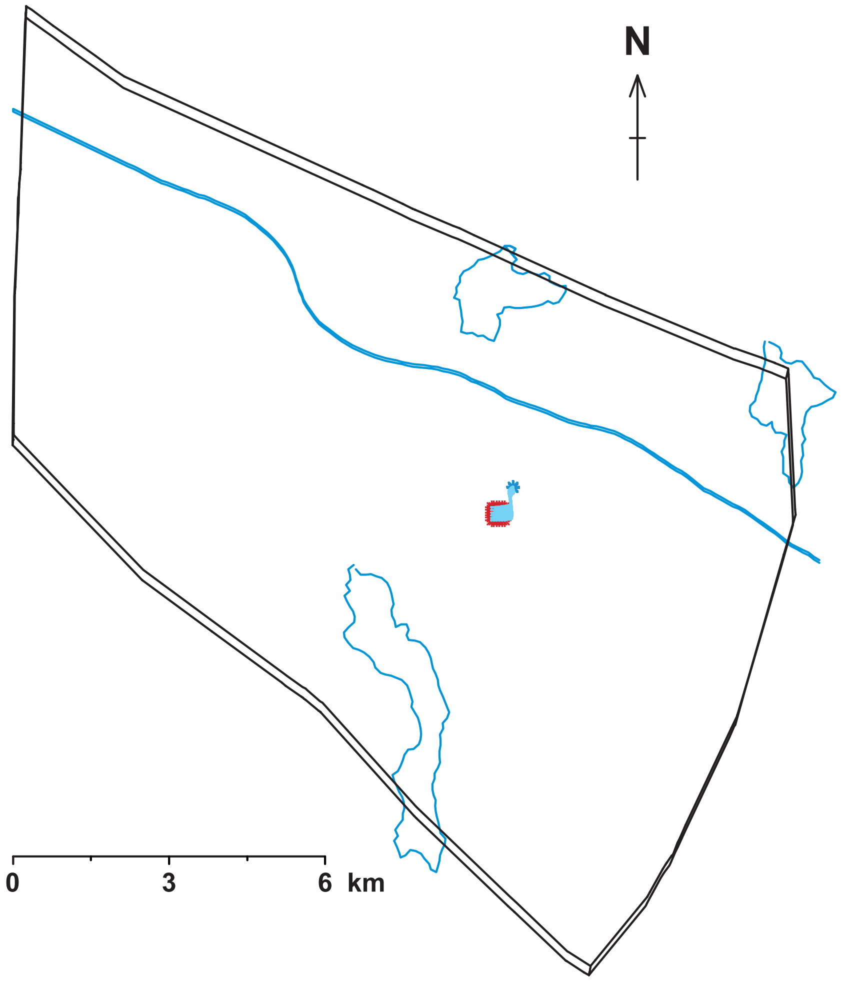

Figure 7-3 *Start positions (red stars) and end positions (blue stars) for 36 particles representing SFL 3-5 at Beberg. The figure also shows the area included in the regional model plus three lakes in the area and a major road.*

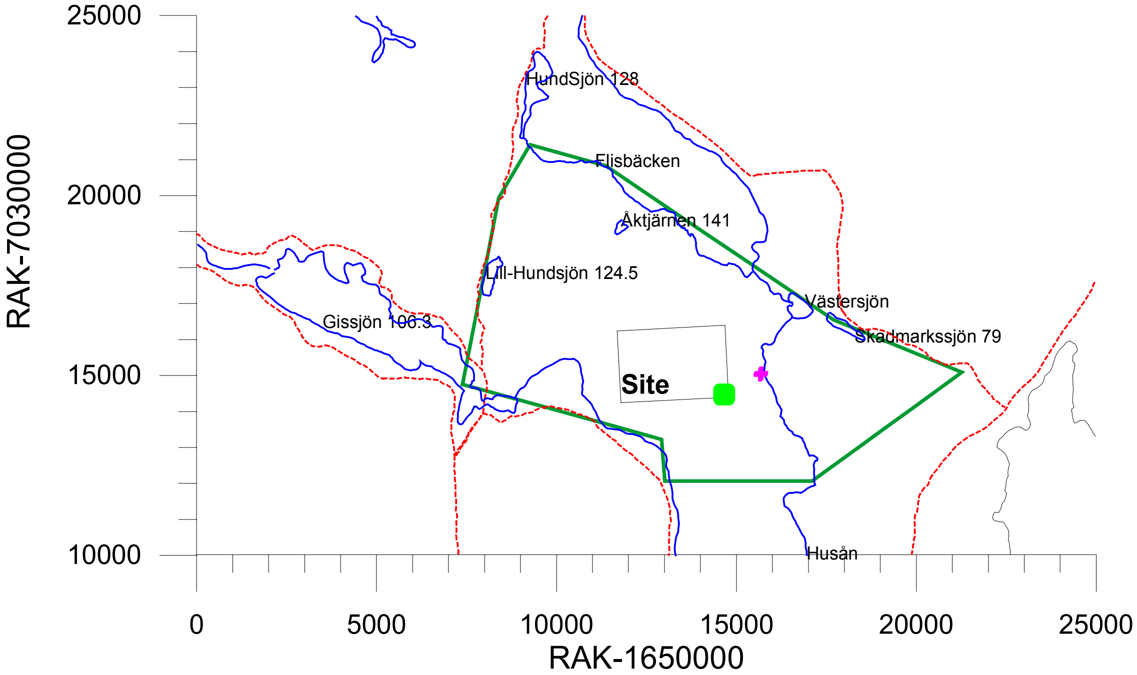

Figure 7-4 *Start and end positions for 36 particles representing SFL 3-5 at Ceberg. Green points show start positions, red cross marks end points and green line marks the boundaries of the model area.*

7.5 Hydrological input data to migration calculations

The results from the different hydrology calculations summarized in the preceding sections have been used to provide input data for the calculations of transport of radionuclides and toxic metals in the near-field barriers in SFL 3-5. The results have also been used for the calculations of migration in the far field and resulting environmental impact. The chosen data and the reasons for these choices are summarized in the following sections (Skagius *et al.*, 1999).

7.5.1 Near-field data

The water flows in the different near-field barriers in SFL 3-5 are calculated as multiples of the regional flow in surrounding rock. The calculations have been carried out for different contrasts in conductivity between the barriers and surrounding rock, and different directions of the regional water flow, see Section 7.3.

The results from the regional hydrology models show that the water flow in the rock at repository depth is largely horizontal on all three sites. The results from the generic modelling of the water flows in the near field show that a horizontal regional flow along SFL 3 and SFL 5 gives the highest flow in these repository parts, while the highest flow in SFL 4 is obtained for a horizontal flow directed perpendicular to SFL 3 and SFL 5. Since the waste in SFL 3 and SFL 5 is much more radioactive than the waste in SFL 4, it has been assumed in the choice of input data that the regional flow is horizontal and directed along SFL 3 and SFL 5 on all three sites. The values of regional water flows chosen to calculate water flows in the near-field barriers are: 10 l/m^{2}y at Aberg, 1 l/m^{2}y at Beberg and 0.1 l/m^{2}y at Ceberg. These values are of the same order of magnitude as the specific flows from the regional models (Table 7-1). This is lower than the calculated mean values at Aberg and Beberg, but we have not striven for complete agreement since the important thing is to show the importance of the size of the groundwater flow.

The total flows (see Table 7-3) have been obtained by combining results from the near-field hydrology modelling with data on the regional water flows. The regional models have been used to obtain the regional water flows. The specific flows (see Table 7-3) are calculated from the total flows and the cross-sections of the different barriers in the flow direction. These specific water flows in the near-field barriers in SFL 3-5 have been used in the calculations of transport of radionuclides and toxic metals in the near field. The values for SFL 4 apply to one of the two parallel tunnel sections.

Table 7-3 Total (m^3/y) and specific (m^3/m^{2}y) water flows in the near-field barriers in SFL 3-5 for a horizontal regional water flow along SFL 3 and SFL 5.

Site, ground-water flow	SFL 3		SFL 4	SFL 5	
	Encapsulation (concrete)	Backfill (gravel)	Backfill[1)] gravel)	Encapsulation (concrete)	Backfill (gravel)
Aberg. 0.01 m/y	0.01 m^3/y 10^{-4} m/y	60 m^3/y 0.31 m/y	240 m^3/y 3.75 m/y	0.01 m^3/y 10^{-4} m/y	62 m^3/y 0.32 m/y
Beberg. 10^{-3} m/y	10^{-3} m^3/y 10^{-5} m/y	6 m^3/y 0.031 m/y	24 m^3/y 0.38 m/y	10^{-3} m^3/y 10^{-5} m/y	6.2 m^3/y 0.032 m/y
Ceberg. 10^{-4} m/y	10^{-4} m^3/y 10^{-6} m/y	0.6 m^3/y 0.0031 m/y	2.4 m^3/y 0.038 m/y	10^{-4} m^3/y 10^{-6} m/y	0.62 m^3/y 0.0032 m/y

1) the flow in one of the two parallel tunnel sections

The results from the near-field model have been chosen in the light of the conductivity of the barriers. The hydraulic conductivities in the rock at repository depth are of the order of 10^{-7} m/s at Aberg, 10^{-8} m/s at Beberg and 10^{-10} m/s at Ceberg, see Chapter 4. The conductivity in the concrete encapsulation in SFL 3 and SFL 5 is set at 10^{-8} m/s and outside, in the crushed rock backfill, to at least 10^{-4} m/s (Skagius *et al.*, 1999). The conductivity in the concrete is chosen to simulate a slightly cracked concrete, see Section 6.7. All of these cases are not dealt with in the calculations of flows in the near field, but this is not necessary either. The calculations have shown that the water flow in the encapsulation is proportional to the contrast in conductivity between the two barriers backfill and encapsulation. With the assumed conductivities, this contrast is 10,000. One condition that must be met is that the backfill is sufficiently permeable, i.e. its conductivity is equal to or greater than 1,000 times that of the rock (Holmén, 1997). This condition is met for all three sites. The flow in the backfill is then independent of its conductivity. The near-field hydraulic calculations deal with a case that is appropriate to use, with a contrast of 10 for the conductivity in encapsulation/rock and 10^5 for backfill/host rock.

As regards SFL 4, a case is dealt with in the near-field hydraulic calculations which corresponds exactly to the situation at Beberg. We have used the same calculation results for Aberg and Ceberg. For Aberg, this may mean that the water flow in the barriers in SFL 3 and SFL 5 are underestimated about 40%, while the water flow in SFL 4 is overestimated by the same order of magnitude. For Ceberg, this means that the flow in the barriers in SFL 3 and SFL 5 is overestimated, while the flow in SFL 4 is not affected (Holmén, 1997). These differences are insignificant in the context and do not affect the conclusions in the report.

7.5.2 Far-field data

To calculate the transport of radionuclides and toxic metals in the far field, data are needed on travel times for groundwater from the repository to the ground surface. To calculate travel times, flow porosities are needed, which are also needed to calculate the surface area in the rock that is available for sorption and matrix diffusion of radionuclides, i.e. the “flow-wetted surface area”. In the light of the results in Table 7-2, an advective travel time for water from the repository to the ground surface of 10 years was chosen for Aberg, 40 years for Beberg and 900 years for Ceberg.

For both Beberg and Ceberg, a flow porosity of 10^{-4} was used to calculate the travel times in Table 7-2 (Hartley and Lindgren, 1997). The same value was used to calculate the flow-wetted surface area at Beberg and Ceberg. The groundwater travel time at Aberg is calculated with a varying flow porosity along the flow paths. The arithmetic mean is $1.3 \cdot 10^{-3}$ (Svensson, 1997b). This value does not overestimate the rock’s capacity to retard radionuclides via sorption and matrix diffusion, and we have therefore used it to calculate the flow-wetted surface area at Aberg (Skagius *et al.*, 1999).

Besides travel times and flow porosities, the results from the hydrology calculations also provide information on which areas on the ground surface are discharge areas for radionuclides from the repository. This information is needed to determine what ecosystem type dominates in these areas and thereby what dose factors should be used for the calculations of dose to man, see Chapter 5.

The classification of the three sites into different ecosystems that was done in SR 97 (Nordlinder *et al.*, 1999) and the discharge areas for groundwater that has passed a repository are shown in Figures 7-5 to 7-7. At Aberg (Figure 7-5), the discharge areas are classified as archipelago and open coast. At Beberg (Figure 7-6), discharge from the repository takes place in an area classified as agricultural land, while at Ceberg (Figure 7-7) the typical ecosystem is classified as peatland. At Beberg, a peat bog is situated immediately adjacent to the agricultural land where the discharge is projected to take place. Since a peat area in general has higher EDFs than an agricultural area, we have chosen to examine the consequence of a discharge to peatland at Beberg.

Figure 7-5 Typical ecosystems and discharge areas at Aberg.

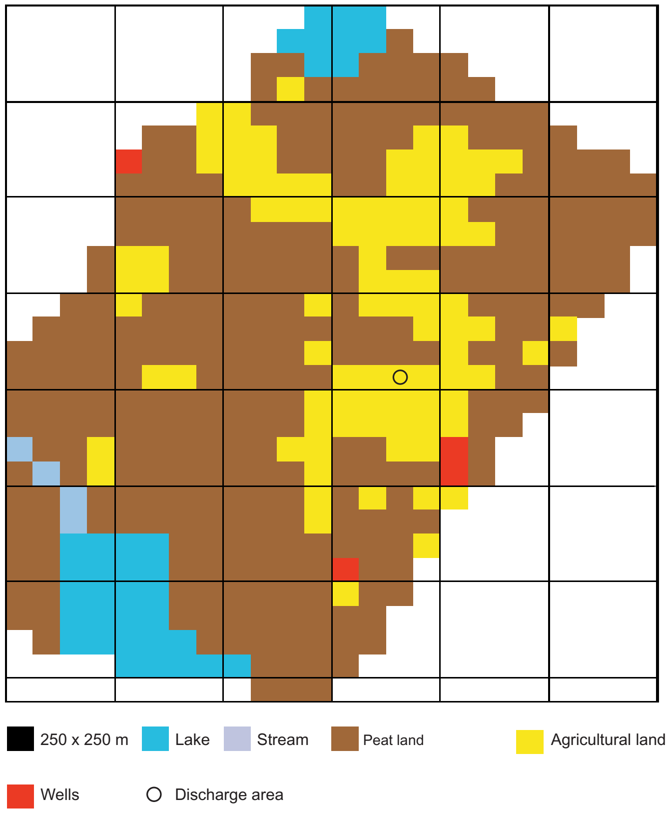

Figure 7-6 Typical ecosystems and discharge area at Beberg.

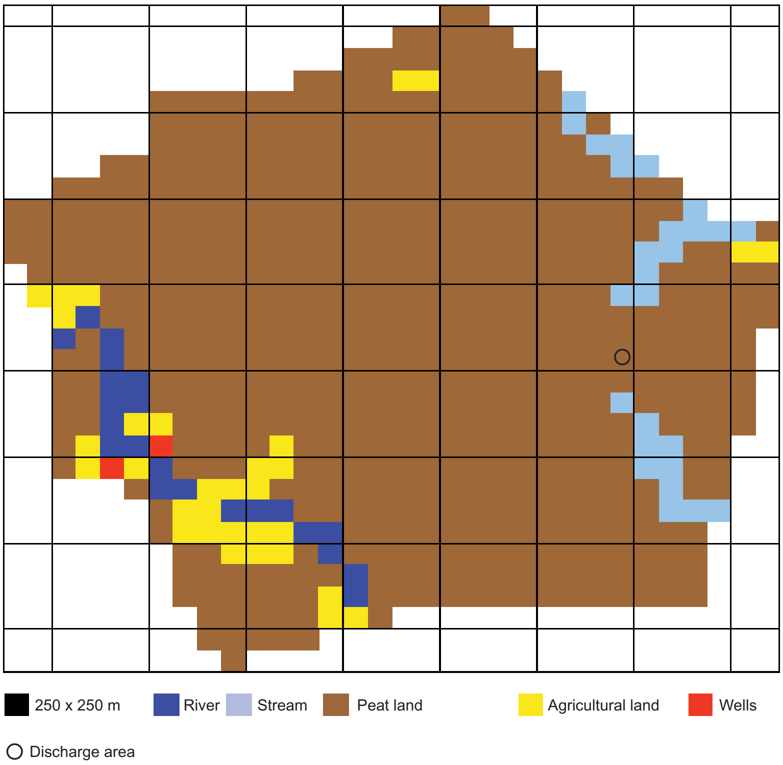

Figure 7-7 Typical ecosystems and discharge area at Ceberg.

7.6 References

Boghammar A, Grundfelt B, Hartley L, 1997. Investigation of the large scale regional hydrogeological situation at Ceberg. SKB Technical Report TR 97-21, Svensk Kärnbränslehantering AB, Stockholm.

Hartley L, Boghammar A, Grundfelt B, 1998. Investigation of the large scale regional hydrogeological situation at Beberg. SKB Technical Report TR 98-24, Svensk Kärnbränslehantering AB, Stockholm.

Hartley L., Lindgren M, 1997. Flow and transport parameters for SFL 3-5. Estimates from regional numerical models for Beberg and Ceberg. Appendix in Skagius *et al.*, 1999.

Holmén J, 1997. On the flow of groundwater in closed tunnels. Generic hydrogeological modelling of nuclear waste repository, SFL 3-5. SKB Technical Report TR 97-10, Svensk Kärnbränslehantering AB, Stockholm.

Nordlinder S, Bergström U, Mathiasson L, 1999. Ecosystem specific dose conversion factors for Aberg, Beberg and Ceberg. SKB Technical Report TR 99-15, Svensk Kärnbränslehantering AB, Stockholm.

Skagius K, Pettersson M, Wiborgh M, Albinsson Y, Holgersson S, 1999. Compilation of data for the analysis of radionuclide migration from SFL 3-5. SKB report R-99-13, Svensk Kärnbränslehantering AB, Stockholm.

Svensson U, 1997a. A regional analysis of groundwater flow and salinity distribution in the Äspö area. SKB Technical Report TR 97-09, Svensk Kärnbränslehantering AB, Stockholm.

Svensson U, 1997b. Flow and transport from SFL 3-5. Estimates from a regional numerical model. Appendix in Skagius *et al.*, 1999.

8 Radionuclide transport

8.1 Introduction

This chapter deals with transport of radionuclides in the near field and geosphere (Pettersson *et al.*, 1999). The different calculation cases dealt with lie within the framework of the reference scenario (Chapter 6). In order to enable the consequences of a calculated release to be analyzed, migration in the geosphere and the biosphere plus dose to man have been calculated. In the reference scenario, the geosphere and biosphere are assumed to be stable and are regarded as far as possible in the same way as in the SR 97 study (SKB, 1999).

Transport of chemotoxic pollutants has also been calculated (Pettersson *et al.*, 1999). These consist of the metals lead, cadmium and beryllium, which are present in a portion of the waste (Chapter 2). An estimate of the consequence of releases of ^{14}C in gaseous form has also been made, and is presented at the end of this chapter.

8.2 Calculation cases and premises

In order to investigate how the water flow in the rock at repository depth affects the transport of radionuclides, three different cases have been treated. The three cases represent three different groundwater flows: 0.1 l/m^{2}y (Ceberg), 1 l/m^{2}y (Beberg) and 10 l/m^{2}y (Aberg), see Section 7.5.

The importance of organic complexing agents is another important subject. For example, alkaline degradation of cellulose in the waste can give rise to isosaccharinic acid (ISA), which is a strong complexing agent. ISA increases the solubility of certain radionuclides. There may also be smaller amounts of other organic materials in the waste with properties similar to ISA. Cellulose and other organic materials are concentrated to SFL 3. To shed light on the influence of organic complexing agents, SFL 3 has been modelled both with and without influence of ISA on radionuclide solubility. The calculation case without ISA influence is referred to in the following as the basic calculation case.

The waste that will be disposed of in SFL 4 contains a great deal of surface-contaminated material. An alternative that has been discussed is to wash off the surface contamination in order to reduce the activity in SFL 4. The activity that is washed off would instead be disposed of in SFL 3. This substantially reduces the activity in SFL 4, but leads to only a very small relative increase of the total activity in SFL 3. Two cases have therefore been dealt with: SFL 4 with and without CRUD. For SFL 4 with CRUD, only the releases from the near field have been calculated.

It is not unusual for the groundwater down at repository depth to have a relatively high salinity. High salinity reduces sorption on minerals for some radionuclides. The geohydrochemical conditions in Beberg are well-suited for testing the importance of salinity. There are two possible compositions of the groundwater, since the boundary between non-saline and saline water lies close to repository depth. This naturally gives us two calculation cases with non-saline (potable) and saline water, but otherwise identical conditions.

All calculation cases are based on the reference scenario and the results of the hydrology calculations, see Chapters 6 and 7. The following premises have therefore been chosen for calculating radionuclide transport in the barriers:

- The time it takes to refill the repository with groundwater after closure is neglected and all repository parts are saturated with water in 2040.
- Waste containers of steel are already fully permeable to water at closure. The corrosion products that eventually form will not affect water flows through the containers or diffusive transport of dissolved components in the water.
- The properties of concrete moulds and concrete enclosures in SFL 3 and SFL 5 do not change with time.
- In all repository parts, the gravel backfill has high permeability to water. Most of the water that flows through SFL 3 and SFL 5 goes through the gravel backfill in the space between the enclosure and the tunnel wall. Any small changes that may occur in the porosity and sorption properties of the fill over long periods of time can be neglected.
- Gas that forms in the repository does not affect water flows or dissolution of radionuclides and chemotoxic pollutants, nor their transport in the water.
- The water flow through SFL 3-5 is horizontal and parallel to SFL 3 and SFL 5 for all three repository sites.
- Water inside the concrete enclosures in SFL 3 and SFL 5, and water inside concrete-grouted waste packages in SFL 4, has a pH of about 12.5 or higher.
- The water in the gravel outside the enclosures in SFL 3 and SFL 5, and outside the waste packages in SFL 4, has the same composition as the groundwater on the three repository sites.
- ^{14}C, ^{36}Cl, ^{93}Zr and ^{93}Mo, which are mainly present in the form of induced activity in metal parts in the waste in SFL 5, are released as the metal parts corrode. ^{59}Ni is also present in the metal parts, but the release of this nuclide is limited by its solubility, see below. 5% of the inventory of ^{3}H and ^{10}Be is immediately accessible for dissolution in the water. The remaining 95% has to decay for 100 years before it can dissolve in the water. The rest of the radionuclides, as well as the toxic pollutants, are immediately accessible for dissolution in the water regardless of where they are located in SFL 3-5.
- The radionuclide ^{59}Ni is solubility-limited in SFL 3 and SFL 5. This also applies to ^{232}Th and ^{238}U in SFL 3, provided the conditions are not affected by ISA. No further reduction in solubility due to isotope dilution has been included.

- Dissolved radionuclides and toxic pollutants are transported out through waste packages and concrete barriers to surrounding rock, both by diffusion and by water flowing through the repository. Sorption in concrete and gravel retards their transport.

- Sorption data for high pH (≥ 12.5) and reducing conditions are chosen in waste packages and enclosures. A saline intruding groundwater gives slightly higher sodium and potassium concentrations in the concrete than a normal concrete pore water, which leads to lower sorption of certain ion exchange-sensitive radionuclides. When sorption data have been available for saline concrete pore water, they have been chosen, e.g. different cesium isotopes. For gravel backfill, sorption data are chosen according to the groundwater on the site in question.

- The influence of the complexing agent isosaccharinic acid (ISA) on sorption is neglected, since its concentration will remain low. However, the possibility that ISA will influence solubility cannot be ruled out, so such a case is also calculated (applies only to SFL 3).

- The groundwater transports radionuclides and toxic pollutants in the far field. Transport is retarded due to diffusion and sorption in the rock matrix. Sorption data are chosen according to the groundwater on the site in question.

8.3 Transport of radionuclides in the near field

8.3.1 Calculation model

If they dissolve in the water in the repository, radionuclides can be transported by diffusion or advection (water flow). This applies to other substances as well, for example toxic metals. The computer code COMP24 is used to calculate the transport of solutes in the near field. First the near field is divided up geometrically into a number of cells or "boxes" (Romero *et al.*, 1995). This resembles the discretization that is done in a finite-difference model for three-dimensional problems.

COMP24 is a further development of COMP23 that is being used in the safety assessment of the deep repository for spent fuel (SKB, 1999). To solve the task it is necessary to be able to calculate transient processes, such as when diffusion has not yet achieved a steady state with constant mass transport. It is also necessary to handle chain decay, i.e. the fact that some radionuclides are transported and simultaneously decay to other radionuclides with other properties.

SFL 3

The groundwater flows mainly through the gravel backfill in SFL 3. The water flow through the concrete structure will be very low, see Chapter 7. Diffusion will therefore be the dominant transport process out of the concrete structure. Once in the tunnel's backfill, dissolved radionuclides can be transported both by diffusion and by flowing water. The SFL 3 tunnel has a plug at both ends, which means that released radionuclides can only leave the tunnel via the rock.

The repository is simplified in the model to facilitate the calculations. The simplifications are conservatively chosen so as not to overrate the importance of the barriers. We take into account sorption in 10 m of backfill at the end of the tunnel, where the flow leaves the tunnel, but e.g. loading zones and concrete plugs (see Figure 3-1) are not included in the calculations. The walls and roof of the tunnel will be covered with shotcrete, but this is not included either. The waste volume to be deposited in SFL 3 is equivalent to about 2/3 of the available storage volume, which is equivalent to 76 m of the length of the concrete structure. We conservatively assume that the waste is collected nearest the part where the water leaves the tunnel. It is assumed that the spaces between concrete moulds and drums are filled with porous concrete and the remaining storage volume inside the concrete structure is filled with a gravel backfill.

In the model, a large number of moulds, with their concrete walls and fill of waste and cement, are grouped into a "waste block". We assume that the radionuclides are evenly distributed in the entire waste block and remain evenly distributed even after some are transported out. Drums deposited in SFL 3 are regarded in the same way as moulds, except that most of the porous concrete poured around them is also included.

From the waste blocks, a diffusive transport is calculated through the porous concrete poured around moulds and drums, see Figure 8-1. The diffusion continues through the concrete structure to the gravel backfill outside. Furthermore, we assume combined transport by advection and diffusion along the tunnel, both in the gravel outside the concrete structure and in the porous concrete around the waste blocks. Sorption in concrete and gravel and the half-life of the radionuclides will limit the outward transport.

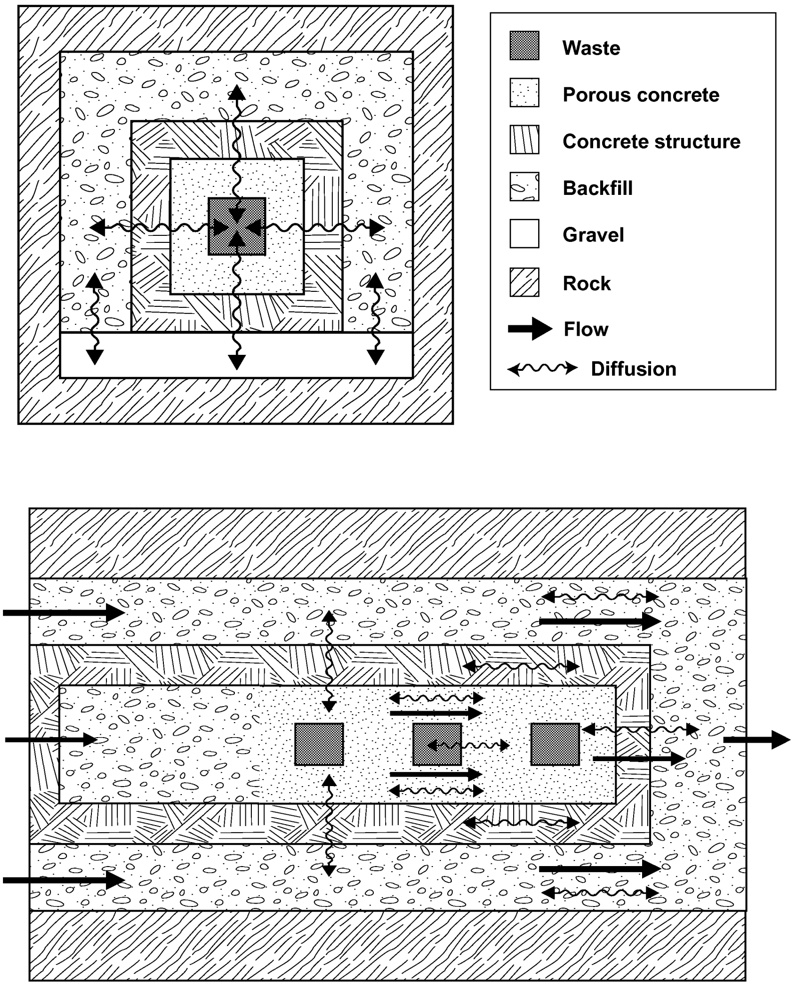

Figure 8-1 Transport pathways for radionuclides in SFL 3. Upper figure: vertical section across main direction of tunnel. Lower figure: horizontal section along main direction of tunnel.

SFL 4

Advection (flow) is most important for solute transport in the SFL 4 tunnel, but an exchange also takes place by diffusion between waste packages and gravel backfill. Released radionuclides can get out into the surrounding rock where the water leaves the tunnel.

Waste from decommissioning of CLAB and the encapsulation plant, i.e. contaminated concrete and metal parts, is the only radioactive waste of importance in SFL 4. Activity in deposited transport casks and containers is insignificant and can be neglected. A large number of waste packages with decommissioning waste are grouped into a “waste block”. This also includes the concrete grout inside the steel case, but the steel walls are not included as a barrier in the calculations.

We assume that the radionuclides are evenly distributed in the entire waste block and remain evenly distributed even after some are transported out. Dissolved radionuclides are transported along the tunnel both by advection and by diffusion, see Figure 8-2. This combined transport takes place both through the waste blocks and through the surrounding gravel backfill. Furthermore, a diffusive transport takes place from waste blocks to gravel. Credit is taken for sorption in waste blocks and gravel, but sorption of radionuclides in the shotcrete and the concrete floor is neglected in the model. Only about 60% of the length of the SFL 4 tunnel contains waste with any significant activity. We conservatively assume that the waste with radionuclides is collected nearest the part where the water leaves the tunnel.

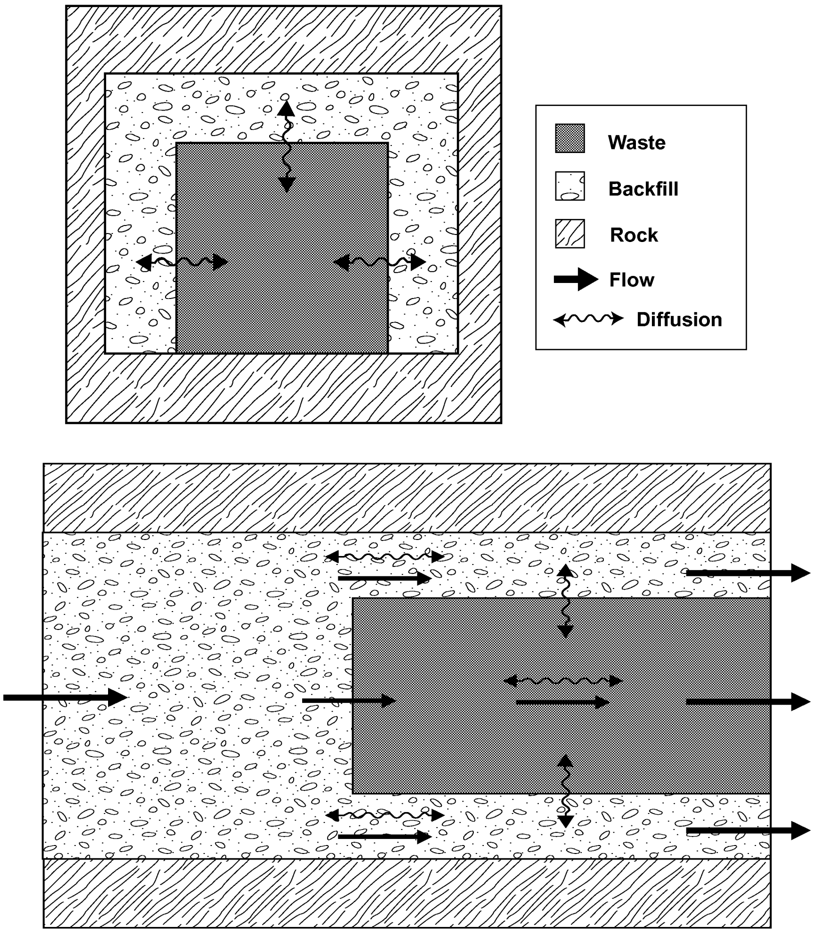

Figure 8-2 Transport pathways for radionuclides in SFL 4. Upper figure: vertical section across main direction of tunnel. Lower figure: horizontal section along main direction of tunnel.

SFL 5

SFL 5 has the same layout as SFL 3, which means that nuclides are transported from waste to surrounding rock in the same way in SFL 5 as in SFL 3. The difference is that SFL 5 contains another type of waste package. The waste, consisting of metal parts, is placed in a long concrete mould with an inner steel cassette, see Chapter 2. Concrete is injected through the lid. Unlike in SFL 3, all waste packages are of a uniform type in SFL 5. The waste volume that is intended for SFL 5 actually exceeds the available storage volume by approximately four percent. We assume in the calculations that the waste can be concentrated so that it will fit into the available space. There are also other possibilities, see Section 3.3.5.

The calculation model for SFL 5 is therefore largely identical to the one for SFL 3. A large number of moulds filled with waste and concrete are grouped into a "waste block" in the calculations. We disregard the steel cassette, i.e. it is not regarded as a barrier. The concrete walls of the mould are, however, taken credit for as a barrier. All empty spaces around the moulds inside the concrete structure are backfilled with porous concrete.

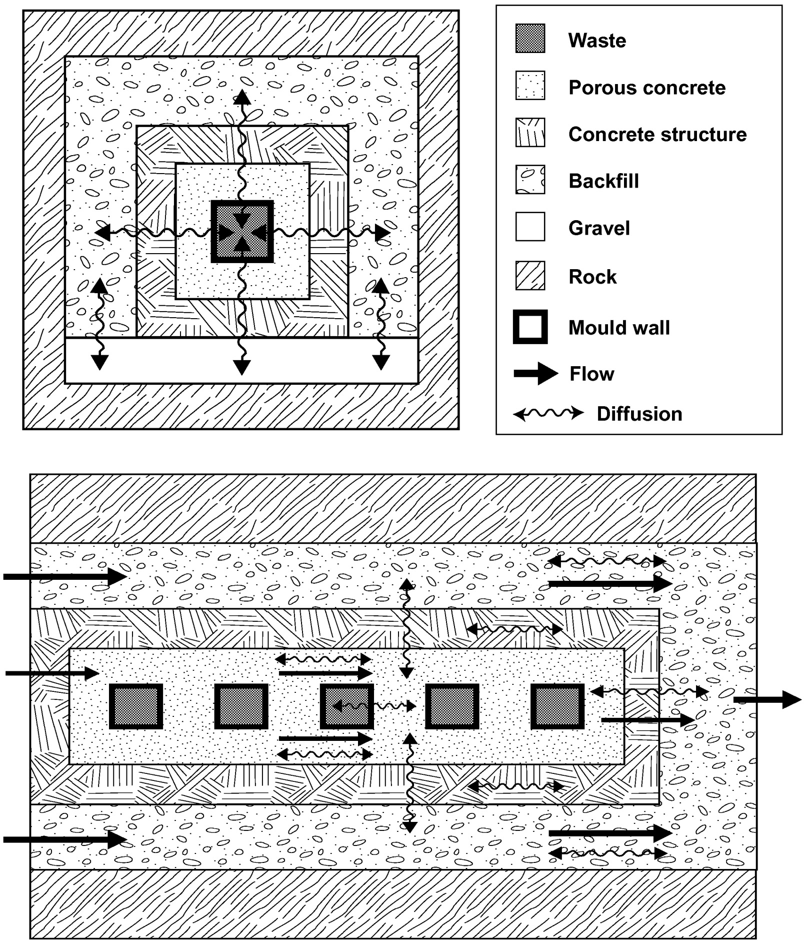

Figure 8-3 Transport pathways for radionuclides in SFL 5. Upper figure: vertical section across main direction of tunnel. Lower figure: horizontal section along main direction of tunnel.

Dissolved radionuclides can diffuse from the waste through the mould walls, through the porous concrete outside the moulds and further through the concrete structure to the gravel backfill outside, see Figure 8-3. Furthermore, we assume combined transport by advection and diffusion along the tunnel, both in the gravel outside the concrete structure and in the porous concrete around the waste blocks. Sorption in concrete and gravel retards the outward transport.

8.3.2 Input data used in the calculations

Water flow

Water flows at repository depth in Aberg, Beberg and Ceberg are dealt with in Chapter 7. The estimated specific water flows that arise in the different near-field barriers of the facility on the different sites have been used in the near-field migration calculations, see Table 7-3.

Quantity of radionuclides in the waste

The radionuclides present in the waste are summarized in Chapter 2 (Table 2-3). Transport in the near field has been calculated for all of these radionuclides. To decide which ones are important to include in further calculations, the releases from the near field are converted to dose. The calculated radiation dose is the dose to an adult person obtained by ingestion of the released radionuclides (IAEA, 1996). It is a purely fictitious case that does not have anything to do with any release scenario, but is used solely to simplify the calculations. For the nuclides that are not included in the further calculations, the calculated radiation dose is low or negligible in comparison with other nuclides (Pettersson *et al.*, 1999). The initial inventory of the remaining nuclides is given in Table 8-1.

Table 8-1 Initial inventory (Bq) of the radionuclides included in the entire calculation chain (year 2040)

Radionuclide	Half-life yrs a)	SFL 3	SFL 4 b)	SFL 5
H-3	12	$3.2\cdot10^{12}$	$1.6\cdot10^{9}$	$2.5\cdot10^{15}$
Be-10	$1.5\cdot10^{6}$	– c)	– c)	$1.4\cdot10^{11}$
$C\text{-}14_{inorg}$	$5.7\cdot10^{3}$	$3.5\cdot10^{13}$	$3.1\cdot10^{7}$	$1.7\cdot10^{14}$
$C\text{-}14_{org}$	$5.7\cdot10^{3}$	$8.2\cdot10^{4}$	– c)	– c)
Cl-36	$3.0\cdot10^{5}$	$2.1\cdot10^{10}$	$2.6\cdot10^{4}$	$2.5\cdot10^{11}$
Co-60	5.3	– c)	$4.1\cdot10^{10}$	– c)
Ni-59	$7.6\cdot10^{4}$	$1.6\cdot10^{14}$	$1.2\cdot10^{8}$	$1.4\cdot10^{15}$
Ni-63	$1.0\cdot10^{2}$	– c)	$1.7\cdot10^{10}$	– c)
Se-79	$1.1\cdot10^{6}$	$4.6\cdot10^{8}$	$5.0\cdot10^{3}$	$4.5\cdot10^{7}$
Sr-90	29	$2.3\cdot10^{12}$	$1.6\cdot10^{8}$	$5.6\cdot10^{11}$
Zr-93	$1.5\cdot10^{6}$	$2.1\cdot10^{10}$	– c)	$2.2\cdot10^{12}$
Nb-94	$2.0\cdot10^{4}$	– c)	$6.6\cdot10^{6}$	–
Mo-93	$4.0\cdot10^{3}$	$2.4\cdot10^{11}$	$2.0\cdot10^{5}$	$1.8\cdot10^{12}$
Tc-99	$2.1\cdot10^{5}$	$5.8\cdot10^{11}$	$6.2\cdot10^{6}$	$3.2\cdot10^{11}$
I-129	$1.6\cdot10^{7}$	$3.4\cdot10^{7}$	$3.7\cdot10^{2}$	$3.4\cdot10^{6}$
Cs-135	$2.3\cdot10^{6}$	$5.7\cdot10^{8}$	– c)	– c)
Cs-137	30	– c)	$1.2\cdot10^{9}$	– c)
Pb-210	22	$2.7\cdot10^{11}$	< 1	< 1
Ra-226	$1.6\cdot10^{3}$	$3.8\cdot10^{11}$	< 1	< 1
Th-229	$7.3\cdot10^{3}$	$1.4\cdot10^{2}$	– c)	– c)
Th-230	$7.5\cdot10^{4}$	$1.8\cdot10^{5}$	< 1	73
Th-232	$1.4\cdot10^{10}$	$1.1\cdot10^{10}$	– c)	– c)
U-233	$1.6\cdot10^{5}$	$3.1\cdot10^{4}$	– c)	– c)
U-234	$2.5\cdot10^{5}$	$7.8\cdot10^{8}$	24	$2.4\cdot10^{5}$
U-236	$2.3\cdot10^{7}$	$8.1\cdot10^{7}$	– c)	– c)
U-238	$4.5\cdot10^{9}$	$4.6\cdot10^{10}$	9.6	$7.5\cdot10^{4}$
Np-237	$2.1\cdot10^{6}$	$1.8\cdot10^{8}$	– c)	– c)
Pu-238	88	$3.7\cdot10^{11}$	$8.9\cdot10^{4}$	$5.9\cdot10^{8}$
Pu-240	$6.6\cdot10^{3}$	$1.8\cdot10^{12}$	– c)	– c)
Pu-241	14	$4.4\cdot10^{12}$	– c)	– c)
Pu-242	$3.7\cdot10^{5}$	$1.2\cdot10^{9}$	72	$5.6\cdot10^{5}$
Am-241	$4.3\cdot10^{2}$	$5.0\cdot10^{12}$	– c)	– c)
Am-242m	$1.4\cdot10^{2}$	$2.0\cdot10^{9}$	$2.4\cdot10^{2}$	$1.6\cdot10^{6}$
Cm-244	18	$4.4\cdot10^{10}$	– c)	– c)
Cm-245	$8.5\cdot10^{3}$	$7.7\cdot10^{7}$	– c)	– c)
Cm-246	$4.7\cdot10^{3}$	$2.1\cdot10^{7}$	1.9	$1.5\cdot10^{4}$

a) Firestone (1998)

b) Assuming decontamination of storage canisters

c) Screened out due to low near-field release

Diffusion, sorption, solubility and corrosion rate

Density, effective diffusivity and porosity are essential physical properties of concrete and gravel that are needed to calculate solute transport in the near field, see Table 8-2. The reasons for the choice of data are given in Skagius *et al.* (1999).

Table 8-2 Physical properties of materials in SFL 3-5.

Material	Bulk density kg/m^3	Effective diffusivity m^2/s	Porosity
Structural concrete	2,295	$1\cdot10^{-11}$	0.15
Porous concrete	1,890	$1\cdot10^{-10}$	0.3
Gravel	1,890	$6\cdot10^{-10}$	0.3

Important chemical parameters are the distribution coefficients (K_d values) for sorption in concrete and gravel, see Table 8-3. In concrete, the pore water in itself has a fairly high concentration of dissolved ions. A saline intruding groundwater can, however, give rise to slightly higher sodium and potassium concentrations in the water in the concrete and thereby a slightly lower sorption of certain ion-exchange sensitive elements, e.g. caesium, when the groundwater is saline. Sorption of other elements can be regarded as being independent of the salinity in the groundwater occurring in Aberg, Beberg and Ceberg. Sorption data for saline conditions in concrete have been chosen where available. We can further assume that the distribution coefficients are the same in both structural and porous concrete. We have used the same sorption data for gravel in the near field as for granite in the far field (Carbol and Engkvist, 1997). Where data are lacking for some of the more unusual nuclides in Table 8-1 we have taken values from the literature, or arrived at K_d values by comparison with chemically analogous nuclides. The reasons for the choice of K_d values are given in Skagius *et al.* (1999).

The composition of the groundwater dominates in the gravel and in the far field, which is of importance for the sorption of radionuclides. We must use other sorption coefficients in the saline groundwaters that can occur in Aberg and Beberg than in non-saline groundwaters for some of the nuclides. The nuclides that primarily sorb by ion exchange, such as Sr, are particularly sensitive.

We conservatively assume that most of the radionuclides dissolve immediately when water enters the waste, regardless of the waste form. The dissolved radionuclides can be sorbed on the concrete and we get a distribution between the two states: dissolved in water and sorbed in concrete (phase distribution). If the radionuclide has very low solubility, the solubility may set an upper limit for the concentration in the water. The values used in the assessment are given in Table 8-4 and discussed in Skagius *et al.* (1999).

Degradation of cellulose in an alkaline environment gives rise to isosaccharinic acid (ISA), which is a strong complexing agent. ISA can form in SFL 3, but the concentration will be low. Experiments show that the low concentration in combination with the relatively long residence times of the radionuclides make the influence of ISA on sorption negligible, but that the solubilities are affected, see Table 8-4 (Skagius *et al.*, 1999).

It is assumed in the assessment of SFL 5 that ^{14}C, ^{36}Cl, ^{93}Zr and ^{93}Mo are released as the metal parts in the waste corrode (Pettersson *et al.,* 1999). The corrosion rate of steel and Zircaloy are assumed to be 1 µm/y and 0.01 µm/y, respectively.

Table 8-3 Distribution coefficients for granite and concrete (K_d m^3/kg).

Element	Concrete	Rock/gravel in saline groundwater	Rock/gravel in non-saline groundwater
H	0	0	0
Be	0.05	0.02	0.1
$C_{inorganic}$	0.2	0.001	0.001
$C_{organic}$	0	0	0
Cl	0.006	0	0
Fe	0.04	0.02	0.1
Co	0.04	0.02	0.1
Ni	0.04	0.02	0.1
Se	0.006	0.001	0.001
Sr	0.001	0.0002	0.01
Zr	0.5	1	1
Nb	0.5	1	1
Mo	0.006	0	0
Tc	0.5	1	1
Pd	0.04	0.01	0.1
Ag	0.001	0.05	0.5
Cd	0.04	0.02	0.1
Sn	0.5	0.001	0.001
Sb	5	2	2
I	0.003	0	0
Cs	0.001	0.05	0.5
Ba	0.007	0.007	0.2
Pm	5	2	2
Sm	5	2	2
Eu	5	2	2
Ho	5	2	2
Pb	0.1	0.02	0.1
Ra	0.05	0.02	0.1
Ac	1	3	3
Th	5	5	5
Pa	5	1	1
U	5	5	5
Np	5	5	5
Pu	5	5	5
Am	1	3	3
Cm	1	3	3

Table 8-4 Upper limits of solubility in concrete pore water (mol/l) (Holgersson *et al.*, 1999).

Element	Solubility	Solubility with ISA
Ni	$1 \cdot 10^{-7}$	$2 \cdot 10^{-7}$
Pm	$9 \cdot 10^{-8}$	$3 \cdot 10^{-5}$
Th	$5 \cdot 10^{-9}$	$7 \cdot 10^{-5}$
U [a)]	$5 \cdot 10^{-9}$	$7 \cdot 10^{-5}$

a) Analogous to Th

8.3.3 Calculated releases from the near field

SFL 3

The release of radionuclides from the near field to SFL 3 has been calculated for the reference scenario on all three sites. The highest release rates were obtained in Aberg, and they occur at two different points in time, see Figure 8-4. The first peak comes after 20–100 years, with a release rate of about $4 \cdot 10^{7}$ Bq/y, and is dominated by ^{3}H (tritium). The second peak reaches roughly as high as the first and occurs after approximately 10^{4}–10^{5} years. The second peak consists mainly of ^{59}Ni. Between these peaks, the total release rate is 10–100 times lower and is dominated by ^{93}Mo, with significant contributions from ^{90}Sr, inorganic ^{14}C and ^{36}Cl. After approximately 10^{6} years, ^{226}Ra and ^{210}Pb dominate. The release curves for these nuclides have two peaks; the first stems from the initial inventory of ^{226}Ra, and the second is obtained when ^{238}U and its daughters ^{234}U and ^{230}Th in the chain 4N+2 decay on the way towards the stable ^{206}Pb.

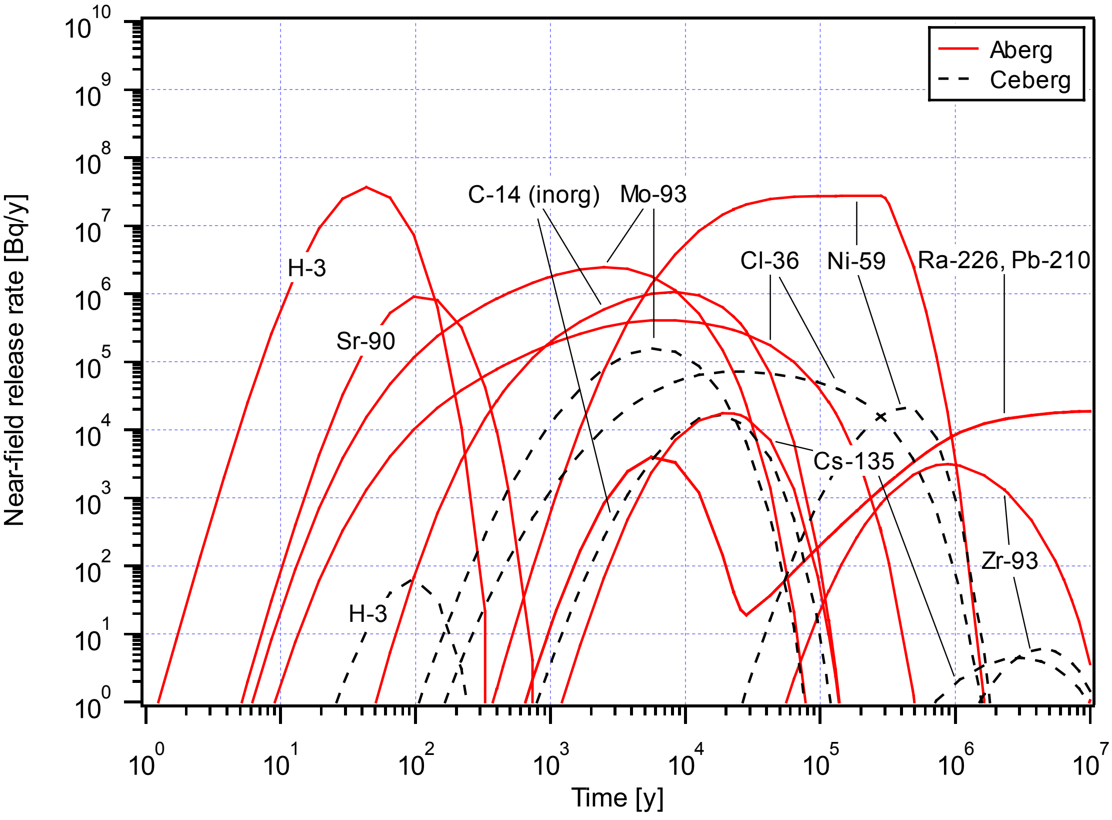

Figure 8-4 Radionuclide release rates from SFR 3 from the near field in Aberg and Ceberg for the basic calculation case within the reference scenario (Beberg, with intermediate release rates, is not included in the figure)

The lowest calculated release rates from SFL 3 are obtained in Ceberg. More or less the same radionuclides dominate, but the release rates of short-lived nuclides such as ^{3}H and ^{90}Sr are much lower than in Aberg. This is particularly clear for ^{90}Sr, whose maximum release rate is 14 orders of magnitude lower in Ceberg than in Aberg. Long-lived nuclides are also affected, but the reduction in release rates is not as marked. Due to radionuclide decay, ^{59}Ni for example does not have as much importance for the total release rate in Ceberg as in Aberg. The biggest release rate in Ceberg is $2{\cdot}10^5$ Bq/y, consists of ^{93}Mo and occurs after 6,000 years.

For some nuclides, such as ^{59}Ni, the difference in release rate between Aberg and Ceberg is a result of a change in both water flow and water composition (salinity). This makes it difficult to determine the influence of the water flow alone. This is, however, possible for e.g. ^{3}H, ^{14}C, ^{36}Cl and ^{93}Mo, since the sorption of these nuclides is not appreciably affected by the salinity of the groundwater. Here we can clearly see that a lower water flow in Ceberg results in lower release rates than in Aberg, see Figure 8-4.

In Beberg, release rates intermediate between those in Aberg (most) and Ceberg (least) are obtained, see Table 8-5. The salinity of the groundwater can vary down at repository depth in Beberg. This makes it necessary to investigate two cases: high and low salinity. This in turn means that Beberg can be used as an example of the importance of salinity. It turns out that saline groundwater leads to release rates that are approximately one order of magnitude higher for ^{135}Cs, and a factor of five higher for ^{59}Ni, ^{210}Pb and ^{226}Ra. The relative difference for ^{90}Sr is even greater. In saline groundwater, ^{90}Sr gives a maximum release rate of $1{\cdot}10^3$ Bq/y, whereas in non-saline groundwater it decays to a large extent, see Table 8-5.

We have also chosen to calculate the transport of radionuclides for a case with influence of ISA in SFL 3. It doesn't make any difference, with one important exception: the solubility of nickel, promethium, uranium and thorium increases. The greatest influence is obtained on the nuclides ^{238}U and ^{232}Th, which are no longer solubility-limited when ISA is present. The release rate of these nuclides is thereby higher, which in turn leads to an increase in the release rate of ^{234}U, ^{230}Th, ^{226}Ra and ^{210}Pb, which come after ^{238}U in the 4N+2 chain. The maximum release rate of nuclides affected by ISA is approximately one order of magnitude higher, except the release rate of ^{59}Ni that increase by a factor of two.

Table 8-5 Maximum release rates of dominant radionuclides from SFL 3's near field and time of occurrence. The calculations apply to the basic case within the reference scenario.

Nuclide	Aberg		Beberg (saline)		Beberg (non-saline)		Ceberg	
	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs
H-3	$4{\cdot}10^7$	$4{\cdot}10^1$	$4{\cdot}10^4$	$1{\cdot}10^2$	$4{\cdot}10^4$	$9{\cdot}10^2$	$6{\cdot}10^1$	$1{\cdot}10^2$
C-14$_{inorg}$	$1{\cdot}10^6$	$8{\cdot}10^3$	$4{\cdot}10^5$	$1{\cdot}10^4$	$4{\cdot}10^5$	$1{\cdot}10^4$	$2{\cdot}10^4$	$2{\cdot}10^4$
Cl-36	$4{\cdot}10^5$	$6{\cdot}10^3$	$3{\cdot}10^5$	$8{\cdot}10^3$	$3{\cdot}10^5$	$8{\cdot}10^3$	$7{\cdot}10^4$	$3{\cdot}10^4$
Ni-59	$3{\cdot}10^7$	$3{\cdot}10^5$	$1{\cdot}10^7$	$3{\cdot}10^5$	$2{\cdot}10^6$	$3{\cdot}10^5$	$2{\cdot}10^4$	$5{\cdot}10^5$
Se-79	$9{\cdot}10^3$	$8{\cdot}10^3$	$5{\cdot}10^3$	$2{\cdot}10^4$	$5{\cdot}10^3$	$2{\cdot}10^4$	$1{\cdot}10^3$	$6{\cdot}10^4$
Sr-90	$9{\cdot}10^5$	$1{\cdot}10^2$	$1{\cdot}10^3$	$2{\cdot}10^2$	$1{\cdot}10^{-5}$	$3{\cdot}10^2$	$9{\cdot}10^{-9}$	$3{\cdot}10^2$
Mo-93	$2{\cdot}10^6$	$2{\cdot}10^3$	$1{\cdot}10^6$	$4{\cdot}10^3$	$1{\cdot}10^6$	$4{\cdot}10^3$	$2{\cdot}10^5$	$6{\cdot}10^3$

SFL 4

There are two different cases for SFL 4. Either the CRUD that contaminates the storage canisters is washed off before deposition, or the canisters are packaged and disposed of as they are (Chapter 2). This gives us two different cases to calculate release rates for: with and without CRUD. Disregarding the radionuclides in the CRUD, SFL 4 contains only a very small quantity of long-lived nuclides. As a result, the total release rate from SFL 4 declines rapidly with time. Washed-off CRUD ends up instead as waste in SFL 3, but there it makes only a negligible relative contribution to the quantity of radionuclides. SFL 4 provides less protection than the other two repository parts, since the water flow there is higher. Furthermore, the assessment of the barriers in SFL 4 is more conservative. We assume, for example, that the waste containers have been degraded so that groundwater flows through the waste.

We start by considering the waste with washed storage canisters, which is the most likely case. Tritium (^{3}H) gives the highest release rate from SFL 4, at both Aberg and Ceberg, see Figure 8-5 and Table 8-6. In Aberg, the maximum release of tritium from SFL 4 is roughly the same as from SFL 3. In Ceberg, most of the ^{3}H in SFL 3 decays before release, but not in SFL 4, which therefore dominates completely with a maximum release of ^{3}H that is four orders of magnitude higher. At longer times, from 40 years onward, the release in Aberg is dominated by ^{63}Ni, followed by ^{59}Ni and ^{99}Tc. Decay reduces the contribution made by ^{63}Ni even more in Ceberg, so the isotope dominates the release for a shorter time than in Aberg. In contrast, ^{59}Ni gives a release in Ceberg that extends over a much longer period of time than in Aberg, due to the fact that ^{59}Ni has decayed completely after on the order of 10^{4} years in Aberg.

The results for Beberg show what the salinity of the groundwater entails for SFL 4 as well, see Table 8-6. The sorption of some nuclides decreases in saline groundwater, which in turn reduces the retardation of their transport. This is particularly clear for ^{90}Sr, which in saline groundwaters in Beberg gives a high release after very short times; the maximum release rate comes after only six years. In non-saline groundwaters, the release of ^{90}Sr is considerably lower during the first ten years. The maximum release rate is obtained after 40 years and is an order of magnitude lower than for saline groundwaters. The maximum release rate of ^{63}Ni, which dominates the near-field release between 100 and 700 years after closure, increases by one order of magnitude in saline waters, while the maximum release rate of ^{59}Ni, which dominates at longer times, is doubled. Note that ^{3}H, which gives the highest release rate from SFL 4 in Beberg, is unaffected by the salinity of the groundwater.

Table 8-6 Maximum release rates of dominant radionuclides from SFL 4's near field and time of occurrence. The calculations apply to the reference scenario and the case without CRUD.

Nuclide	Aberg		Beberg (saline)		Beberg (non-saline)		Ceberg	
	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs
H-3	$8\cdot10^{7}$	1	$6\cdot10^{6}$	3	$6\cdot10^{6}$	3	$6\cdot10^{5}$	3
C-14_{inorg}	$2\cdot10^{3}$	$2\cdot10^{2}$	$3\cdot10^{2}$	$9\cdot10^{2}$	$3\cdot10^{2}$	$9\cdot10^{2}$	$3\cdot10^{1}$	$1\cdot10^{3}$
Co-60	$4\cdot10^{4}$	$2\cdot10^{1}$	$1\cdot10^{3}$	$2\cdot10^{1}$	2	$3\cdot10^{1}$	$2\cdot10^{-1}$	$3\cdot10^{1}$
Ni-59	$3\cdot10^{4}$	$1\cdot10^{3}$	$3\cdot10^{3}$	$8\cdot10^{3}$	$2\cdot10^{3}$	$2\cdot10^{3}$	$1\cdot10^{2}$	$2\cdot10^{3}$
Ni-63	$1\cdot10^{6}$	$1\cdot10^{2}$	$9\cdot10^{4}$	$1\cdot10^{2}$	$1\cdot10^{4}$	$2\cdot10^{2}$	$1\cdot10^{3}$	$2\cdot10^{2}$
Sr-90	$1\cdot10^{6}$	4	$1\cdot10^{5}$	6	$1\cdot10^{4}$	$4\cdot10^{1}$	$1\cdot10^{3}$	$4\cdot10^{1}$
Mo-93	$4\cdot10^{2}$	$3\cdot10^{1}$	$6\cdot10^{1}$	$2\cdot10^{2}$	$6\cdot10^{1}$	$2\cdot10^{2}$	7	$2\cdot10^{2}$
Cs-137	$2\cdot10^{5}$	$6\cdot10^{1}$	$5\cdot10^{3}$	$1\cdot10^{2}$	1	$2\cdot10^{2}$	$1\cdot10^{-1}$	$2\cdot10^{2}$

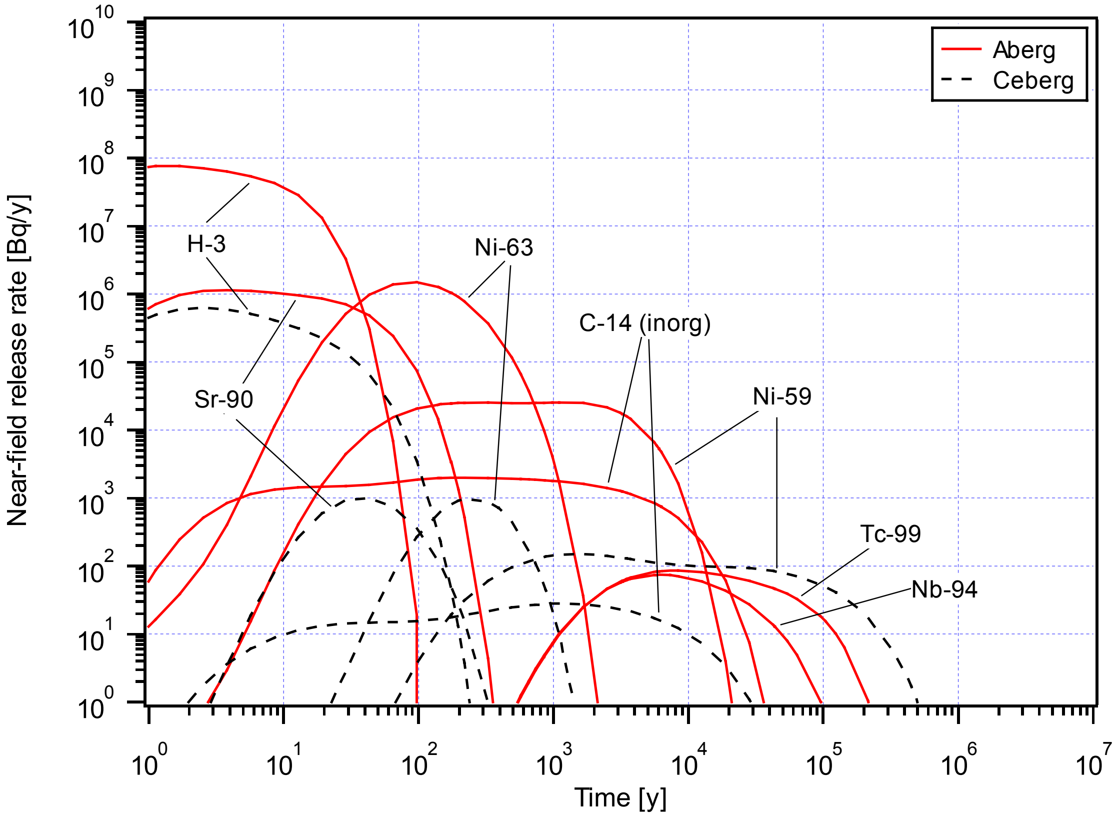

Figure 8-5 Radionuclide release rates from SFL 4 from the near field in Aberg and Ceberg (reference scenario and case without CRUD).

If the storage canisters are not washed off before deposition in SFL 4, the inventory of some radionuclides will be considerably higher. For example, the quantity of ^{14}C, ^{90}Sr, ^{93}Mo and ^{99}Tc increases by three orders of magnitude, ^{59}Ni and ^{63}Ni increase by two orders of magnitude, while the quantity of ^{3}H triples. Due to the larger quantity of ^{90}Sr, this radionuclide gives the maximum release rate from SFL 4 in Aberg and with saline groundwaters in Beberg, see Table 8-7.

Table 8-7 Maximum release rates of dominant radionuclides from SFL 4's near field and time of occurrence. The calculations apply to the reference scenario and the case with CRUD.

Nuclide	**Aberg**		**Beberg (saline)**		**Beberg (non-saline)**		**Ceberg**	
	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs
H-3	$2\cdot10^{8}$	1	$2\cdot10^{7}$	3	$2\cdot10^{7}$	3	$2\cdot10^{6}$	3
C-14_{inorg}	$2\cdot10^{6}$	$2\cdot10^{2}$	$2\cdot10^{5}$	$9\cdot10^{2}$	$2\cdot10^{5}$	$9\cdot10^{2}$	$2\cdot10^{4}$	$1\cdot10^{3}$
Co-60	$3\cdot10^{7}$	$2\cdot10^{1}$	$8\cdot10^{5}$	$2\cdot10^{1}$	$2\cdot10^{3}$	$3\cdot10^{1}$	$1\cdot10^{2}$	$3\cdot10^{1}$
Ni-59	$6\cdot10^{6}$	$1\cdot10^{3}$	$7\cdot10^{5}$	$8\cdot10^{3}$	$4\cdot10^{5}$	$2\cdot10^{3}$	$3\cdot10^{4}$	$2\cdot10^{3}$
Ni-63	$5\cdot10^{8}$	$9\cdot10^{1}$	$3\cdot10^{7}$	$1\cdot10^{2}$	$4\cdot10^{6}$	$2\cdot10^{2}$	$3\cdot10^{5}$	$2\cdot10^{2}$
Sr-90	$1\cdot10^{9}$	4	$1\cdot10^{8}$	6	$1\cdot10^{7}$	$4\cdot10^{1}$	$1\cdot10^{6}$	$4\cdot10^{1}$
Mo-93	$3\cdot10^{5}$	$3\cdot10^{1}$	$4\cdot10^{4}$	$2\cdot10^{2}$	$4\cdot10^{4}$	$2\cdot10^{2}$	$4\cdot10^{3}$	$2\cdot10^{2}$
Cs-137	$2\cdot10^{8}$	$6\cdot10^{1}$	$6\cdot10^{6}$	$8\cdot10^{1}$	$2\cdot10^{3}$	$2\cdot10^{2}$	$1\cdot10^{2}$	$2\cdot10^{2}$

SFL 5

SFL 5 is the repository part that contains the most radionuclides and gives the highest release rates from the near field. The nuclides that dominate the release are ^{3}H at $8\cdot10^{8}$ Bq/y (Aberg), ^{59}Ni at $1\cdot10^{7}$ Bq/y (Beberg, saline groundwater), ^{93}Mo at $3\cdot10^{6}$ Bq/y (Beberg, non-saline waters) and ^{36}Cl at $4\cdot10^{5}$ Bq/y (Ceberg), see Figure 8-6 and Table 8-8.

The great similarity between SFL 5 and SFL 3 explains why the time of release of a given radionuclide is the same for SFL 3 and SFL 5. The difference in release between SFL 3 and SFL 5 reflects above all differences in the radionuclide inventory. The nuclides that dominate the release during the period up to 10^{6} years are the same as in SFL 3. In calculations that extend towards extremely long times, ^{93}Zr or ^{10}Be dominates.

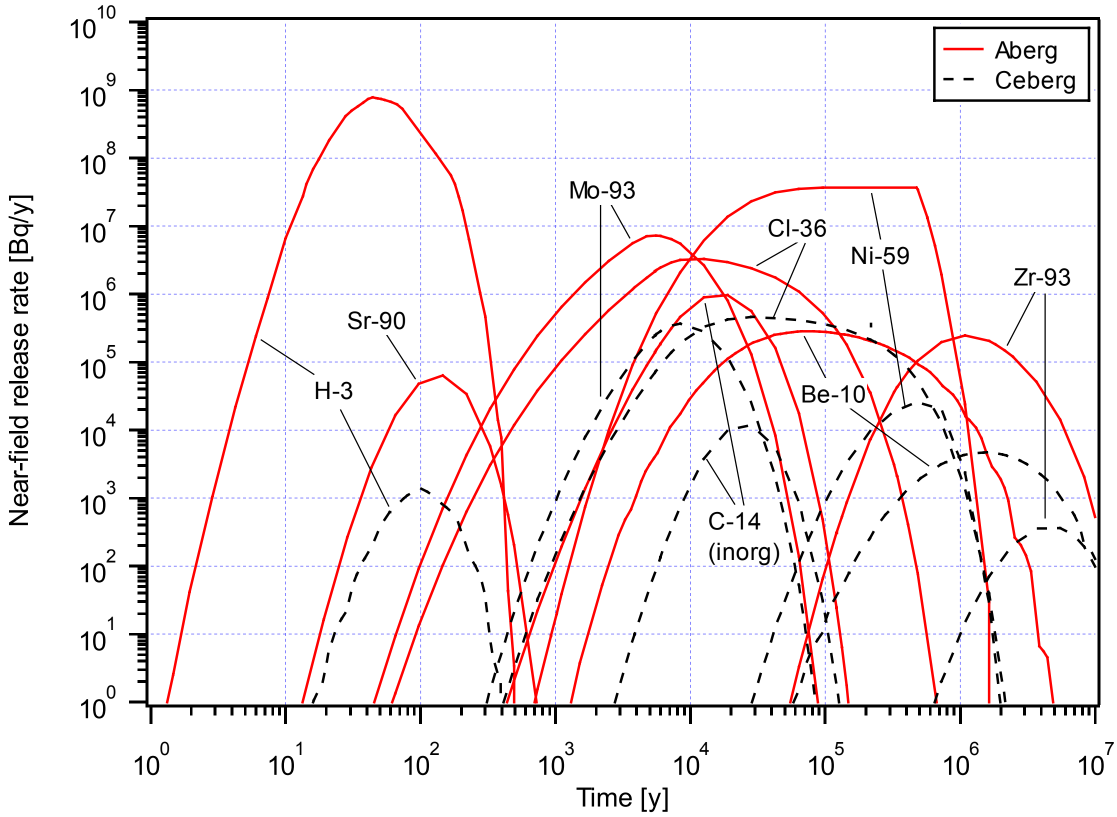

Figure 8-6 Radionuclide release rates from SFL 5 from the near field in Aberg and Ceberg (reference scenario).

The influence of groundwater salinity on radionuclide release from SFL 5 in Beberg is the same as from SFL 3 in Beberg, which was discussed earlier in this chapter. It is the same nuclides that are affected, with two exceptions: ^{135}Cs is not included in the calculations for SFL 5, whereas ^{10}Be is. ^{10}Be is so long-lived that variations in sorption have little importance for the maximum release rate, see Table 8-8.

Table 8-8 Maximum release rates of dominant radionuclides from SFL 5's near field and time of occurrence. The calculations apply to the reference scenario.

Nuclide	Aberg		Beberg (saline)		Beberg (non-saline)		Ceberg	
	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs
H-3	$8\cdot10^{8}$	$4\cdot10^{1}$	$1\cdot10^{6}$	$9\cdot10^{1}$	$1\cdot10^{6}$	$9\cdot10^{1}$	$1\cdot10^{3}$	$1\cdot10^{2}$
Be-10	$3\cdot10^{5}$	$7\cdot10^{4}$	$1\cdot10^{5}$	$2\cdot10^{5}$	$7\cdot10^{4}$	$5\cdot10^{5}$	$5\cdot10^{3}$	$2\cdot10^{6}$
C-14$_{inorg}$	$1\cdot10^{6}$	$2\cdot10^{4}$	$3\cdot10^{5}$	$2\cdot10^{4}$	$3\cdot10^{5}$	$2\cdot10^{4}$	$1\cdot10^{4}$	$3\cdot10^{4}$
Cl-36	$3\cdot10^{6}$	$1\cdot10^{4}$	$2\cdot10^{6}$	$1\cdot10^{4}$	$2\cdot10^{6}$	$1\cdot10^{4}$	$5\cdot10^{5}$	$3\cdot10^{4}$
Ni-59	$4\cdot10^{7}$	$1\cdot10^{5}$	$1\cdot10^{7}$	$5\cdot10^{5}$	$2\cdot10^{6}$	$5\cdot10^{5}$	$3\cdot10^{4}$	$5\cdot10^{5}$
Sr-90	$6\cdot10^{4}$	$1\cdot10^{2}$	$1\cdot10^{2}$	$2\cdot10^{2}$	$8\cdot10^{-7}$	$3\cdot10^{2}$	$7\cdot10^{-10}$	$3\cdot10^{2}$
Zr-93	$2\cdot10^{5}$	$1\cdot10^{6}$	$2\cdot10^{4}$	$3\cdot10^{6}$	$2\cdot10^{4}$	$3\cdot10^{6}$	$4\cdot10^{2}$	$6\cdot10^{6}$
Mo-93	$7\cdot10^{6}$	$5\cdot10^{3}$	$3\cdot10^{6}$	$6\cdot10^{3}$	$3\cdot10^{6}$	$6\cdot10^{3}$	$4\cdot10^{5}$	$8\cdot10^{3}$

8.4 Transport of radionuclides in the far field

8.4.1 Calculation model

Dissolved radionuclides released from the near field can be transported by the water flow in the rock (advection). The water moves in open fractures. A solute can be retarded by sorption on the minerals on the fracture surfaces, or by diffusing into the microfissures in the rock and being sorbed there. The latter phenomenon is called matrix diffusion and is the most important cause of retention during radionuclide transport in the far field. The computer code FARF31 is used to calculate the transport of dissolved radionuclides or other substances in the far field (Norman and Kjellbert, 1990). It is the same code that is used in the safety assessment of the deep repository for spent fuel (SKB, 1999). The calculations are based on a one-dimensional model for transport along a single flow path in the rock. The model takes into account the hydrological properties advection and dispersion along the flow path, as well as chain decay. The model also naturally handles matrix diffusion. The results of the calculations are given in the form of release rates (Bq/y) from the geosphere to the biosphere.

8.4.2 Input data used in the calculations

Transport of radionuclides in the far field has been calculated for a selection of the nuclides found in the repository. The radionuclides that already make only a negligible contribution to the dose after the near field are not included, see Section 8.3.2. Besides release rates for radionuclides from the near field, data needed to calculate transport in the far field include effective diffusivities for different nuclides inside the rock matrix, distribution coefficients and a number of parameters that describe the properties of the rock. These include the advective travel time for groundwater in the rock, Peclet number, flow-wetted surface area (based on volume of flowing water), matrix porosity and maximum penetration depth, see Table 8-9. The advective travel time in the rock and the flow-wetted surface area are based on the results from the regional hydrology model, see Section 7.5.2. Other parameter values are the same as those used in the safety assessment of the deep repository for spent fuel (Andersson, 1999). A more exhaustive description of the choice of far-field data is provided in Skagius *et al.* (1999).

Table 8-9 Compilation of parameter values needed to calculate radionuclide transport in the geosphere.

Parameter	Aberg	Beberg	Ceberg
Advective travel time (yrs)	10	40	900
Peclet number (–)	10	10	10
Flow-wetted surface area (m^2/m^3) [a)]	770	10,000	1,000
Matrix porosity (–) [b)]	0.005	0.005	0.005
Maximum penetration depth (m)	2	2	20

[a)] Based on volume of flowing water

[b)] Due to anion exclusion, the porosity for inorganic ^{14}C, ^{36}Cl and ^{129}I is reduced by one order of magnitude in non-saline waters

Chosen data for effective diffusivities D_e (Ohlsson and Neretnieks, 1997) and distribution coefficients K_d (Carbol and Engkvist, 1997) are given in Table 8-10. To as great an extent as possible, we have chosen the same values as in the safety assessment of the deep repository for spent fuel (designated “best estimate” in (SKB, 1999). Data are, however, lacking for several species that are of interest solely for SFL 3-5. In such cases, D_e is determined using the method described by Ohlsson and Neretnieks (1997). Lacking K_d have mainly be taken from data in the literature. In cases where such data are also lacking, we have estimated the sorption coefficients by means of comparisons with chemically analogous species (Skagius *et al.*, 1999).

D_e and K_d values corresponding to high ionic strengths are used for Aberg, while values for low ionic strengths are used for Ceberg. Both non-saline and saline waters occur in Beberg, so both sets of values are used.

Table 8-10 Effective diffusivities D_e and distribution coefficients K_d in the geosphere for different nuclides and different ionic strengths (saline and non-saline water).

Element	D_e (m^2/s)		K_d (m^3/kg)	
	High ionic strength	Low ionic strength	High ionic strength	Low ionic strength
H	$1.0 \cdot 10^{-13}$	$1.0 \cdot 10^{-13}$	0	0
Be	$2.4 \cdot 10^{-14}$	$2.4 \cdot 10^{-14}$	0.02	0.1
C_{inorg}	$5.0 \cdot 10^{-14}$	$5.0 \cdot 10^{-15}$	0.001	0.001
C_{org}	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	0	0
Cl	$8.3 \cdot 10^{-14}$	$8.3 \cdot 10^{-15}$	0	0
Co	$2.9 \cdot 10^{-14}$	$2.9 \cdot 10^{-14}$	0.02	0.1
Ni	$2.8 \cdot 10^{-14}$	$2.8 \cdot 10^{-14}$	0.02	0.1
Se	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	0.001	0.001
Sr	$3.3 \cdot 10^{-14}$	$3.3 \cdot 10^{-13}$	0.0002	0.01
Zr	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	1	1
Nb	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	1	1
Mo	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	0	0
Tc [a)]	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	1	1
Cd	$3.0 \cdot 10^{-14}$	$3.0 \cdot 10^{-14}$	0.02	0.1
I	$8.3 \cdot 10^{-14}$	$8.3 \cdot 10^{-15}$	0	0
Cs	$8.8 \cdot 10^{-14}$	$8.8 \cdot 10^{-13}$	0.05	0.5
Pb	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	0.02	0.1
Ra	$3.7 \cdot 10^{-14}$	$3.7 \cdot 10^{-14}$	0.02	0.1
Th	$6.3 \cdot 10^{-15}$	$6.3 \cdot 10^{-15}$	5	5
U [a)]	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	5	5
Np [a)]	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	5	5
Pu	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	5	5
Am	$4.0 \cdot 10^{-14}$	$4.0 \cdot 10^{-14}$	3	3

a) Values for Tc(IV), U(IV) and Np(IV), respectively.

8.4.3 Calculated releases from the far field

The results of the calculations of radionuclide transport in the far field based on the reference scenario are presented in this section. In FARF31, the lower limit for the release rate from the far field is set at $1 \cdot 10^{-3}$ Bq/y. Anything less than this is regarded as zero. The maximum release rate from the far field for some nuclides falls below this limit. In result tables with maximum release rates from the far field, this is indicated as $< 1 \cdot 10^{-3}$ Bq/y. The time when the maximum release rate occurs is not specified.

SFL 3

The release of radionuclides from the far field in Aberg has been calculated, see Figure 8-7. The source of the release is SFL 3, with premises chosen from the basic case in the reference scenario, i.e. no influence of ISA on solubilities in the near field. Due to the short advective travel time for the groundwater in Aberg, the releases in the far field differ very little from the releases from the near field. Consequently, the same nuclides dominate the release even after transport through all the rock. The maximum release rate comes from ^{59}Ni and is $2 \cdot 10^{7}$ Bq/y, after about 10^{5} years.

In Ceberg, on the other hand, the flow is very low. Most short-lived nuclides decay before leaving the near field, and the process of decay continues in the far field, where the advective travel time is as long as 900 years. Long-lived nuclides are also greatly affected, for example ^{59}Ni. In the case of other long-lived nuclides such as ^{36}Cl, ^{93}Mo and ^{129}I, the release is retarded but nevertheless reaches roughly the same maximum rate. The radionuclide ^{36}Cl gives rise to the maximum release rate from the far field in Ceberg: $7 \cdot 10^{4}$ Bq/y.

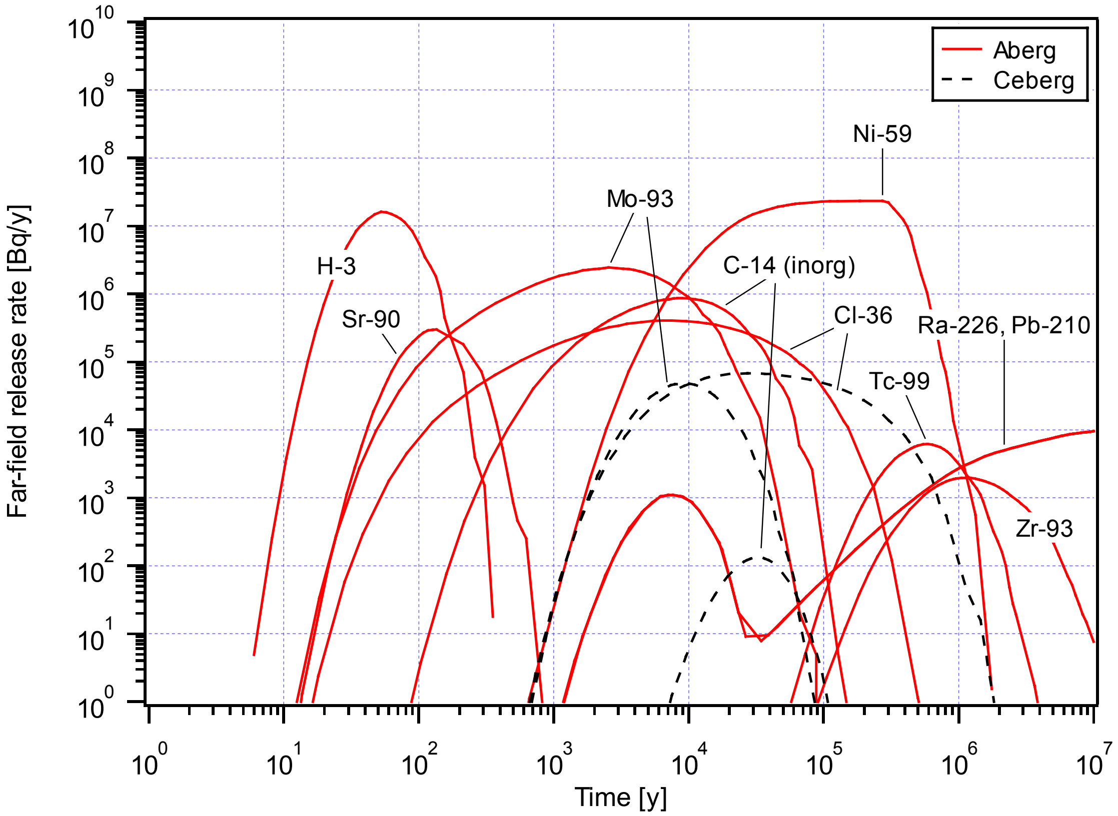

Figure 8-7 Radionuclide release rates from SFL 3 from the far field in Aberg and Ceberg for the basic calculation case (i.e. no influence of ISA on solubilities in the near field).

In Beberg we again have two calculation cases, depending on the salinity we assume in the groundwater. The advective travel times for the groundwater in Beberg are only slightly longer than in Aberg, see Table 8-9. However this, together with a larger flow-wetted surface area in Beberg, is long enough for ^{3}H, which dominates the near-field release during the first 100 years, and ^{90}Sr to decay to a large extent. The importance of ^{59}Ni also declines, despite the fact that it is one of the radionuclides that gives rise to the highest release rate from the near field. The maximum release rate of ^{59}Ni from the far field is two orders of magnitude lower than the maximum calculated release rate from the near field. This is true provided the groundwater is saline. If the groundwater is non-saline, the difference is four orders of magnitude, thanks to better sorption in the rock. In the case with saline groundwater in Beberg, ^{59}Ni dominates the release in the long-term perspective 10^5–10^6 years. Otherwise it is the nuclides ^{36}Cl and ^{93}Mo that give the highest release rates from the far field (10^5–10^6 Bq/y at most). Chlorine and molybdenum form anions (chloride and molybdate ions) that are highly mobile in the rock, as a consequence of which they are not retarded sufficiently during transport and don't have time to decay in the rock. The same applies to ^{129}I, but inorganic ^{14}C has a short enough half-life to be affected by the advective travel time despite its being considered highly mobile (carbonate).

The radionuclides ^{36}Cl, ^{129}I and inorganic ^{14}C, which occur as anions, have a more difficult time penetrating into the microfissures in the rock than cations. The phenomenon is called anion exclusion. Saline groundwater reduces this effect and thereby increases the effective diffusivity of ^{36}Cl, ^{129}I and inorganic ^{14}C in the rock, see

Table 8-10. As a consequence, such radionuclides are retarded more by matrix diffusion in saline groundwaters. This doesn't mean so much in Beberg for ^{36}Cl and ^{129}I, since they have such extremely long half-lives, but the release of inorganic ^{14}C decreases more if the groundwater is saline, see Table 8-11.

The effective diffusivity of strontium and cesium, which occur as cations, is higher in non-saline than in saline groundwaters. This is due to the phenomenon of surface diffusion, which gives the ions a higher diffusivity than they would otherwise have. Saline groundwater counteracts sorption of Sr and Cs and thereby reduces the importance of surface diffusion. We thus get a double negative effect of saline groundwater on the transport of Sr and Cs: both matrix diffusion and sorption are impaired. However, the calculations show that both ^{90}Sr and ^{135}Cs decay to insignificant levels during transport in the far field in Beberg, even if the groundwater is saline, see Table 8-11.

Table 8-11 Maximum release rates of SFL 3's radionuclides from the far field and time of occurrence. The calculations apply to the reference scenario.

Nuclide	**Aberg**		**Beberg (saline)**		**Beberg (non-saline)**		**Ceberg**	
	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs
H-3	$2\cdot10^{7}$	$5\cdot10^{1}$	$1\cdot10^{1}$	$1\cdot10^{2}$	$1\cdot10^{1}$	$1\cdot10^{2}$	$< 1\cdot10^{-3}$	–
C-14$_{\text{inorg}}$	$9\cdot10^{5}$	$9\cdot10^{3}$	$7\cdot10^{2}$	$3\cdot10^{4}$	$3\cdot10^{4}$	$2\cdot10^{4}$	$1\cdot10^{2}$	$3\cdot10^{4}$
C-14$_{\text{org}}$	$6\cdot10^{1}$	$2\cdot10^{2}$	$1\cdot10^{1}$	$2\cdot10^{3}$	$1\cdot10^{1}$	$2\cdot10^{3}$	1	$7\cdot10^{3}$
Cl-36	$4\cdot10^{5}$	$7\cdot10^{3}$	$3\cdot10^{5}$	$1\cdot10^{4}$	$3\cdot10^{5}$	$1\cdot10^{4}$	$7\cdot10^{4}$	$3\cdot10^{4}$
Ni-59	$2\cdot10^{7}$	$3\cdot10^{5}$	$4\cdot10^{4}$	$4\cdot10^{5}$	$1\cdot10^{2}$	$7\cdot10^{5}$	$2\cdot10^{-3}$	$1\cdot10^{6}$
Se-79	$8\cdot10^{3}$	$9\cdot10^{3}$	$6\cdot10^{2}$	$1\cdot10^{5}$	$6\cdot10^{2}$	$1\cdot10^{5}$	$1\cdot10^{2}$	$5\cdot10^{5}$
Sr-90	$3\cdot10^{5}$	$1\cdot10^{2}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Zr-93	$2\cdot10^{3}$	$1\cdot10^{6}$	$3\cdot10^{-2}$	$9\cdot10^{6}$	$3\cdot10^{-2}$	$9\cdot10^{6}$	$< 1\cdot10^{-3}$	–
Mo-93	$2\cdot10^{6}$	$3\cdot10^{3}$	$7\cdot10^{5}$	$4\cdot10^{3}$	$7\cdot10^{5}$	$5\cdot10^{3}$	$5\cdot10^{4}$	$9\cdot10^{3}$
Tc-99	$6\cdot10^{3}$	$6\cdot10^{5}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
I-129	$1\cdot10^{3}$	$4\cdot10^{3}$	$8\cdot10^{2}$	$6\cdot10^{3}$	$8\cdot10^{2}$	$6\cdot10^{3}$	$2\cdot10^{2}$	$2\cdot10^{4}$
Cs-135	$8\cdot10^{3}$	$3\cdot10^{4}$	4	$2\cdot10^{6}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
U-236	8	$9\cdot10^{6}$	$4\cdot10^{-3}$	$8\cdot10^{7}$	$4\cdot10^{-3}$	$9\cdot10^{7}$	$< 1\cdot10^{-3}$	–
Th-232	9	$7\cdot10^{8}$	4	$1\cdot10^{9}$	4	$1\cdot10^{9}$	1	$4\cdot10^{9}$
Np-237	3	$4\cdot10^{6}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
U-233	3	$4\cdot10^{6}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Th-229	4	$4\cdot10^{6}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
U-238	$3\cdot10^{1}$	$1\cdot10^{9}$	7	$2\cdot10^{9}$	7	$2\cdot10^{9}$	1	$5\cdot10^{9}$
U-234	$3\cdot10^{1}$	$3\cdot10^{8}$	7	$2\cdot10^{9}$	7	$2\cdot10^{9}$	1	$5\cdot10^{9}$
Th-230	$3\cdot10^{1}$	$9\cdot10^{7}$	7	$2\cdot10^{9}$	7	$2\cdot10^{9}$	1	$5\cdot10^{9}$
Ra-226	$1\cdot10^{4}$	$7\cdot10^{7}$	$2\cdot10^{3}$	$2\cdot10^{9}$	$3\cdot10^{2}$	$2\cdot10^{9}$	$5\cdot10^{1}$	$5\cdot10^{9}$
Pb-210	$1\cdot10^{4}$	$7\cdot10^{7}$	$2\cdot10^{3}$	$2\cdot10^{9}$	$3\cdot10^{2}$	$2\cdot10^{9}$	$5\cdot10^{1}$	$5\cdot10^{9}$

SFL 4

The release of radionuclides from the far field has been calculated with the assumption that CRUD has been washed off of the storage canisters. In Aberg, the releases of non-sorbing and low-sorbing nuclides from the near field to SFL 4 come almost immediately, see Figure 8-5. The advective travel time for the groundwater in the geosphere is short in Aberg, but still sufficient to delay the time of the maximum release rate of e.g. ^{3}H and ^{90}Sr by several decades, see Figure 8-8. The maximum release rate (Bq/y) for ^{3}H and ^{90}Sr is reduced by a factor of 4 thanks to the far field. The release rate of ^{63}Ni is reduced by a factor of 50. The maximum release rate of the long-lived nuclide ^{59}Ni is only affected insignificantly, but occurs approximately 2,000 years later.

The longer advective travel time and/or the rock's greater capacity to sorb nuclides in Beberg and in Ceberg makes the far field a better barrier than in Aberg. The only nuclides that give rise to release rates in excess of 1 Bq/y in Beberg are ^{3}H, inorganic ^{14}C (non-saline groundwater), ^{36}Cl, ^{59}Ni (saline groundwater) and ^{93}Mo, see Table 8-12. In Ceberg, only ^{93}Mo gives a release rate in excess of 1 Bq/y.

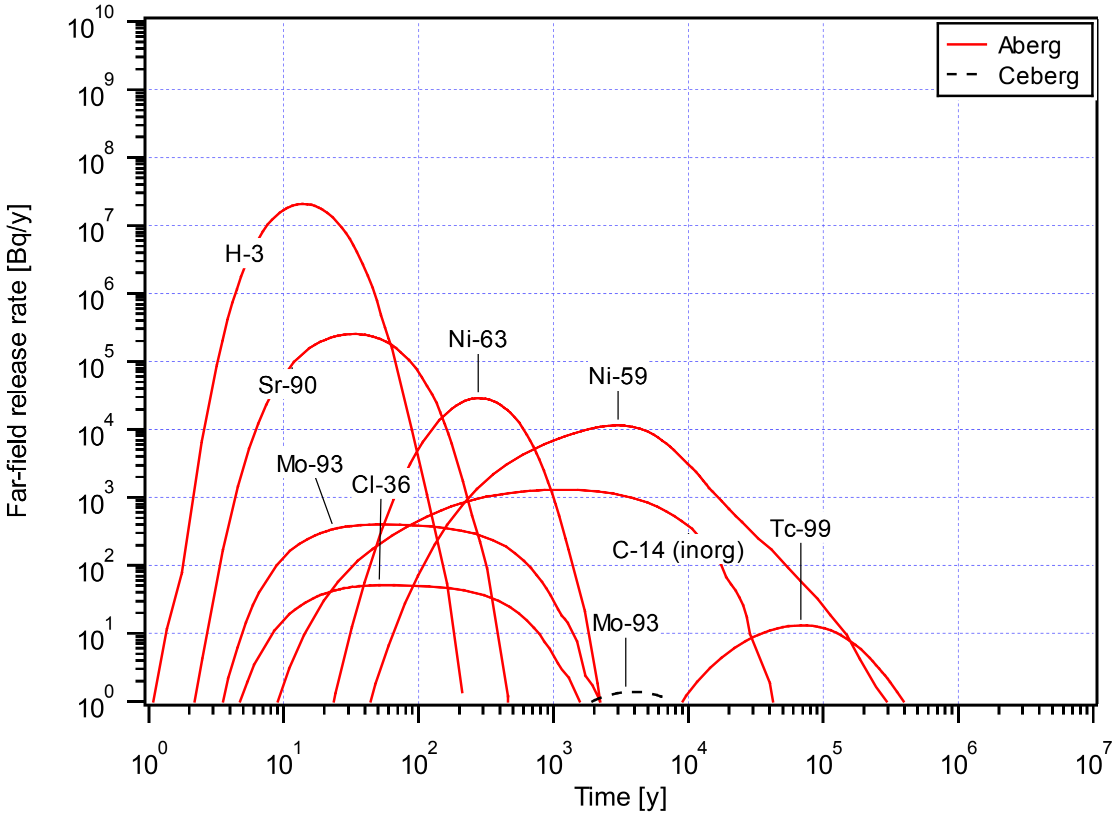

Figure 8-8 Radionuclide release rates from SFL 4 from the far field in Aberg and Ceberg (inventory not including CRUD).

Table 8-12 Maximum release rates of SFL 4's radionuclides from the far field and time of occurrence. The calculations apply to the reference scenario and the case without CRUD.

Nuclide	Aberg		Beberg (saline)		Beberg (non-saline)		Ceberg	
	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs
H-3	$2\cdot10^{7}$	$1\cdot10^{1}$	$6\cdot10^{2}$	$6\cdot10^{1}$	$6\cdot10^{2}$	$6\cdot10^{1}$	$< 1\cdot10^{-3}$	–
C-14_{inorg}	$1\cdot10^{3}$	$1\cdot10^{3}$	$3\cdot10^{-1}$	$2\cdot10^{4}$	$1\cdot10^{1}$	$9\cdot10^{3}$	$1\cdot10^{-1}$	$2\cdot10^{4}$
Cl-36	$5\cdot10^{1}$	$6\cdot10^{1}$	3	$2\cdot10^{3}$	7	$4\cdot10^{2}$	$8\cdot10^{-1}$	$5\cdot10^{3}$
Co-60	1	$5\cdot10^{1}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Ni-59	$1\cdot10^{4}$	$3\cdot10^{3}$	2	$2\cdot10^{5}$	$1\cdot10^{-2}$	$3\cdot10^{5}$	$< 1\cdot10^{-3}$	–
Ni-63	$3\cdot10^{4}$	$3\cdot10^{2}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Se-79	5	$3\cdot10^{2}$	$1\cdot10^{-2}$	$5\cdot10^{4}$	$1\cdot10^{-2}$	$5\cdot10^{4}$	$< 1\cdot10^{-3}$	–
Sr-90	$3\cdot10^{5}$	$3\cdot10^{1}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Nb-94	4	$3\cdot10^{4}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Mo-93	$4\cdot10^{2}$	$5\cdot10^{1}$	$2\cdot10^{1}$	$1\cdot10^{3}$	$2\cdot10^{1}$	$1\cdot10^{3}$	1	$4\cdot10^{3}$
Tc-99	$1\cdot10^{1}$	$7\cdot10^{4}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
I-129	1	$5\cdot10^{1}$	$7\cdot10^{-2}$	$1\cdot10^{3}$	$2\cdot10^{-1}$	$4\cdot10^{2}$	$2\cdot10^{-2}$	$4\cdot10^{3}$
Cs-137	1	$2\cdot10^{2}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
U-238	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
U-234	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Th-230	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Ra-226	$3\cdot10^{-3}$	$3\cdot10^{5}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Pb-210	$3\cdot10^{-3}$	$3\cdot10^{5}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–

SFL 5

There are great similarities between the calculated release rates from SFL 3 and SFL 5. An exception is ^{10}Be, which is projected to give a dominant release from SFL 5 in Beberg and Ceberg after approximately 10^{6} years, see Figure 8-9. In Aberg it is instead ^{93}Zr from SFL 5 that dominates after the same length of time.

The total far-field release rate from SFL 5 is, with the exception of a few short periods of time, consistently slightly higher than from SFL 3 and much higher than from SFL 4. This is true on all three sites.

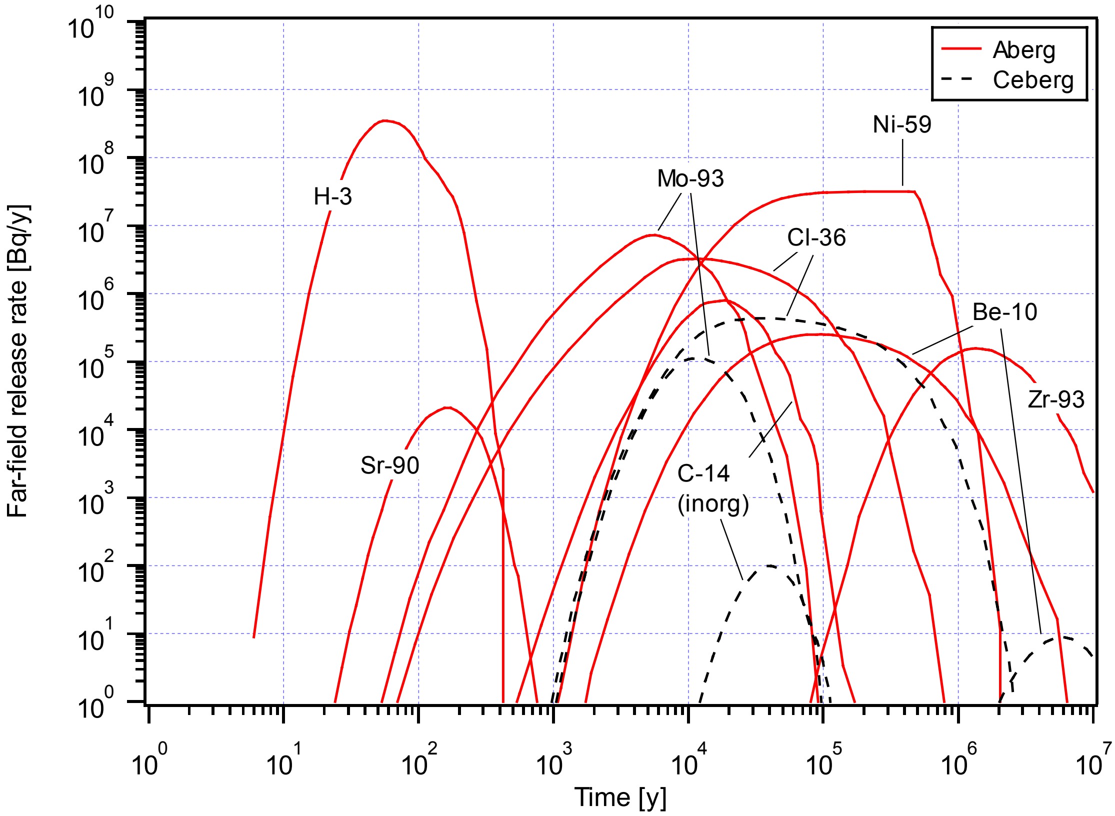

Figure 8-9 Radionuclide release rates from SFL 5 from the far field in Aberg and Ceberg.

Table 8-13 Maximum release rates of SFL 5's radionuclides from the far field and time of occurrence. The calculations apply to the reference scenario.

Nuclide	Aberg		Beberg (saline)		Beberg (non-saline)		Ceberg	
	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs	Maximum release rate Bq/y	Time of maximum release yrs
H-3	$3\cdot10^{8}$	$6\cdot10^{1}$	$3\cdot10^{2}$	$1\cdot10^{2}$	$3\cdot10^{2}$	$1\cdot10^{2}$	$< 1\cdot10^{-3}$	–
Be-10	$3\cdot10^{5}$	$1\cdot10^{5}$	$1\cdot10^{4}$	$1\cdot10^{6}$	$1\cdot10^{3}$	$2\cdot10^{6}$	9	$6\cdot10^{6}$
C-14$_{inorg}$	$8\cdot10^{5}$	$2\cdot10^{4}$	$6\cdot10^{2}$	$4\cdot10^{4}$	$2\cdot10^{4}$	$3\cdot10^{4}$	$1\cdot10^{2}$	$4\cdot10^{4}$
Cl-36	$3\cdot10^{6}$	$1\cdot10^{4}$	$2\cdot10^{6}$	$2\cdot10^{4}$	$2\cdot10^{6}$	$2\cdot10^{4}$	$4\cdot10^{5}$	$3\cdot10^{4}$
Ni-59	$3\cdot10^{7}$	$4\cdot10^{5}$	$5\cdot10^{4}$	$6\cdot10^{5}$	$1\cdot10^{2}$	$8\cdot10^{5}$	$2\cdot10^{-3}$	$1\cdot10^{6}$
Se-79	$7\cdot10^{2}$	$1\cdot10^{4}$	$5\cdot10^{1}$	$2\cdot10^{5}$	$5\cdot10^{1}$	$2\cdot10^{5}$	8	$6\cdot10^{5}$
Sr-90	$2\cdot10^{4}$	$2\cdot10^{2}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Zr-93	$2\cdot10^{5}$	$1\cdot10^{6}$	2	$9\cdot10^{6}$	2	$9\cdot10^{6}$	$< 1\cdot10^{-3}$	–
Mo-93	$7\cdot10^{6}$	$6\cdot10^{3}$	$2\cdot10^{6}$	$8\cdot10^{3}$	$2\cdot10^{6}$	$8\cdot10^{3}$	$1\cdot10^{5}$	$1\cdot10^{4}$
Tc-99	$2\cdot10^{3}$	$6\cdot10^{5}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
I-129	$1\cdot10^{2}$	$4\cdot10^{3}$	$6\cdot10^{1}$	$1\cdot10^{4}$	$6\cdot10^{1}$	$8\cdot10^{3}$	$1\cdot10^{1}$	$2\cdot10^{4}$
U-238	$1\cdot10^{-3}$	$1\cdot10^{7}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
U-234	$1\cdot10^{-3}$	$1\cdot10^{7}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Th-230	$1\cdot10^{-3}$	$1\cdot10^{7}$	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–	$< 1\cdot10^{-3}$	–
Ra-226	$3\cdot10^{-1}$	$1\cdot10^{7}$	$7\cdot10^{-3}$	$3\cdot10^{8}$	$1\cdot10^{-3}$	$4\cdot10^{8}$	$< 1\cdot10^{-3}$	–
Pb-210	$3\cdot10^{-1}$	$1\cdot10^{7}$	$7\cdot10^{-3}$	$3\cdot10^{8}$	$1\cdot10^{-3}$	$3\cdot10^{8}$	$< 1\cdot10^{-3}$	–

8.5 Summary of calculation results for near field and far field

8.5.1 Near field

The maximum release rate from the near field of dose-dominant nuclides (see further Section 8.6.2) and certain other nuclides that are affected by conditions on the site are compiled in Table 8-14. The conclusions that can be drawn from the calculations regarding radionuclide transport in the near field are summarized below.

- The highest near-field release rates for all radionuclides and repository parts are obtained for a siting in Aberg, since Aberg has the highest water flow in the near field.

- The influence of the size of the water flow is greatest on the release of short-lived, low- and non-sorbing radionuclides such as ^{3}H and ^{90}Sr. A more or less direct proportionality to the flow change is obtained for the release of ^{3}H from SFL 4. In SFL 3 and SFL 5, where the concrete structure influences the outward transport, the influence of a reduced flow on the release rate of ^{3}H is even greater at the flows that apply for Aberg, Beberg and Ceberg.

- The radionuclides that dominate the dose for the releases from SFL 3 and SFL 5 are ^{14}C, ^{93}Mo and ^{36}Cl. For these radionuclides as well, it can be seen how the release rate from the near field decreases with decreasing water flows in the near field. However, the flow must be reduced to that prevailing in Ceberg before any obvious changes occurs for ^{93}Mo and ^{36}Cl.

- The differences in the results from Beberg based on non-saline and saline groundwaters demonstrate the importance of the greater sorption in surrounding gravel fill which many cations have in non-saline compared with saline water. A pronounced effect is seen on ^{90}Sr, but ^{59}Ni in SFL 3 and SFL 5 and ^{60}Co and ^{137}Cs in SFL 4 are also affected.

- Some waste types in SFL 3 contain cellulose, which can form isosaccharinic acid, ISA, through alkaline degradation. This complexing agent is not expected to exert any influence on sorption, but the solubility of polyvalent nuclides might increase. This means that uranium and thorium are no longer solubility-limited in SFL 3, leading to an increased release of their daughters.

- The waste to be deposited in SFL 4 contains a lot of material with surface contamination, CRUD. In calculating environmental impact, it has been assumed that the storage canisters from CLAB are decontaminated. Calculations performed with CRUD show that the near-field release of ^{3}H might triple and ^{90}Sr could increase by a factor of one thousand if the storage canisters are not decontaminated before disposal in SFL 4. The near-field release of ^{93}Mo and ^{36}Cl might also increase by a factor of one thousand, but is still lower than the release rate from SFL 3 and SFL 5.

Table 8-14 Release rate from near field of dose-dominant radionuclides and certain radionuclides that are affected by conditions on the three sites.

Site	Groundwater	SFL 3		SFL 4		SFL 5	
		Nuclide	Bq/y	Nuclide	Bq/y	Nuclide	Bq/y
Aberg	saline	H-3	$4\cdot10^{7}$	H-3	$8\cdot10^{7}$	H-3	$8\cdot10^{8}$
		C-14	$1\cdot10^{6}$	Sr-90	$1\cdot10^{6}$	C-14	$1\cdot10^{6}$
		Cl-36	$4\cdot10^{5}$			Cl-36	$3\cdot10^{6}$
		Ni-59	$3\cdot10^{7}$			Ni-59	$4\cdot10^{7}$
		Sr-90	$9\cdot10^{5}$			Mo-93	$7\cdot10^{6}$
		Mo-93	$2\cdot10^{6}$				
Beberg	saline	H-3	$4\cdot10^{4}$	H-3	$6\cdot10^{6}$	H-3	$1\cdot10^{6}$
		C-14	$4\cdot10^{5}$	Sr-90	$1\cdot10^{5}$	C-14	$3\cdot10^{5}$
		Cl-36	$3\cdot10^{5}$			Cl-36	$2\cdot10^{6}$
		Ni-59	$1\cdot10^{7}$			Ni-59	$1\cdot10^{7}$
		Sr-90	$1\cdot10^{3}$			Mo-93	$3\cdot10^{6}$
		Mo-93	$1\cdot10^{6}$				
Beberg	non-saline	H-3	$4\cdot10^{4}$	H-3	$6\cdot10^{6}$	H-3	$1\cdot10^{6}$
		C-14	$4\cdot10^{5}$	Sr-90	$1\cdot10^{4}$	C-14	$3\cdot10^{5}$
		Cl-36	$3\cdot10^{5}$			Cl-36	$2\cdot10^{6}$
		Ni-59	$2\cdot10^{6}$			Ni-59	$2\cdot10^{6}$
		Sr-90	<1			Mo-93	$3\cdot10^{6}$
		Mo-93	$1\cdot10^{6}$				
Ceberg	non-saline	H-3	60	H-3	$6\cdot10^{5}$	H-3	$1\cdot10^{3}$
		C-14	$2\cdot10^{4}$	Sr-90	$1\cdot10^{3}$	C-14	$1\cdot10^{4}$
		Cl-36	$7\cdot10^{4}$			Cl-36	$5\cdot10^{5}$
		Ni-59	$2\cdot10^{4}$			Ni-59	$3\cdot10^{4}$
		Sr-90	<1			Mo-93	$4\cdot10^{5}$
		Mo-93	$2\cdot10^{5}$				

8.5.2 Far field

The maximum release rate from the far field of dose-dominant nuclides (see further Section 8.6.2) and certain other nuclides that are affected by conditions on the site are compiled in Table 8-15. The conclusions that can be drawn from the calculations regarding radionuclide transport in the far field are summarized below.

- The highest release rates from the far field are obtained for Aberg, which has the shortest groundwater residence times. There is very little time for radioactive decay in the far field here, which is why there is a release of the relatively short-lived low- and non-sorbing radionuclides ^{3}H (12 years) and ^{90}Sr (29 years).

- Long groundwater residence times are important for radionuclides that sorb in the far field. Retardation in the rock can drastically reduce the release. For example, the release rate of ^{59}Ni from the near field is of the same order of magnitude in Aberg and Beberg (case with saline groundwater), whereas the release rate from the far field is nearly a factor of one thousand lower for Beberg.

Table 8-15 Release rate from far field of dose-dominant radionuclides and certain radionuclides that are affected by the far-field conditions on the three sites.

Site	Groundwater	SFL 3		SFL 4		SFL 5	
		Nuclide	Bq/y	Nuclide	Bq/y	Nuclide	Bq/y
Aberg	saline	H-3	$2\cdot10^{7}$	H-3	$2\cdot10^{7}$	H-3	$3\cdot10^{8}$
		C-14	$9\cdot10^{5}$	Sr-90	$3\cdot10^{5}$	C-14	$8\cdot10^{5}$
		Cl-36	$4\cdot10^{5}$	Others	$\leq3\cdot10^{4}$	Cl-36	$3\cdot10^{6}$
		Ni-59	$2\cdot10^{7}$			Ni-59	$3\cdot10^{7}$
		Sr-90	$3\cdot10^{5}$			Mo-93	$7\cdot10^{6}$
		Mo-93	$2\cdot10^{6}$				
Beberg	saline	H-3	10	H-3	$6\cdot10^{2}$	H-3	$3\cdot10^{2}$
		C-14	$7\cdot10^{2}$	Sr-90	<1	C-14	$6\cdot10^{2}$
		Cl-36	$3\cdot10^{5}$	Others	≤3	Cl-36	$2\cdot10^{6}$
		Ni-59	$4\cdot10^{4}$			Ni-59	$5\cdot10^{4}$
		Sr-90	<1			Mo-93	$2\cdot10^{6}$
		Mo-93	$7\cdot10^{5}$				
Beberg	non-saline	H-3	10	H-3	$6\cdot10^{2}$	H-3	$3\cdot10^{2}$
		C-14	$3\cdot10^{4}$	Sr-90	<1	C-14	$2\cdot10^{4}$
		Cl-36	$3\cdot10^{5}$	Others	≤7	Cl-36	$2\cdot10^{6}$
		Ni-59	$1\cdot10^{2}$			Ni-59	$1\cdot10^{2}$
		Sr-90	<1			Mo-93	$2\cdot10^{6}$
		Mo-93	$7\cdot10^{5}$				
Ceberg	non-saline	H-3	<1	H-3	<1	H-3	<1
		C-14	$1\cdot10^{2}$	Sr-90	<1	C-14	$1\cdot10^{2}$
		Cl-36	$7\cdot10^{4}$	Others	≤1	Cl-36	$4\cdot10^{5}$
		Ni-59	<1			Ni-59	<1
		Sr-90	<1			Mo-93	$1\cdot10^{5}$
		Mo-93	$5\cdot10^{4}$				

- The groundwater residence time in the rock and the composition of this water are of little importance for the radionuclides that are non-sorbing and long-lived (^{36}Cl and ^{93}Mo). These nuclides give rise to high release rates on all the sites (Table 8-15) and also give the highest dose in Beberg and Ceberg.

- A slightly higher release rate of ^{93}Mo than ^{36}Cl is obtained in Aberg and Beberg. The reverse is true in Ceberg, due to the longer advective travel time which allows ^{93}Mo ($4\cdot10^{3}$ yrs) to decay more than ^{36}Cl ($3\cdot10^{5}$ yrs).

- The results for the different cases with non-saline and saline groundwater in Beberg show what importance the differences in salinity have for transport in the far field. Sorption in saline water decreases for those nuclides that sorb mainly by ion exchange. This effect is clearly apparent for ^{59}Ni, whose release rate is about one hundred times lower in non-saline water than in saline water.

- Anions such as Cl^-, I^- and HCO_3^- are retarded more in saline than in non-saline waters due to lower diffusion in the pores of the rock matrix when the water is non-saline (anion exclusion). The half-life of the radionuclides is of importance, and in Beberg the effect is most noticeable for ^{14}C, which has the shortest half-life.

8.6 Calculation of dose

8.6.1 Model and input data used in the calculations

The model that has been used to calculate transport of radionuclides in the biosphere and to determine the resulting dose to man has been described in Chapter 5. It is the same calculation model as that used for SR 97 (SKB, 1999).

The typical ecosystems for the release from SFL 3-5 are archipelago and open coast in Aberg, agricultural land in Beberg and peatland in Ceberg. We also shed light on a release to a nearby peatland in Beberg, see Section 7.5.2. The mean values of ecosystem-specific dose conversion factors for all typical ecosystems considered are compiled in Table 8-16.

Table 8-16 Mean values of ecosystem-specific dose conversion factors (Sv/Bq) according to Nordlinder *et al.* (1999).

Radio-nuclide	Aberg Open coast	Aberg Archipelago	Beberg Agricultural land	Beberg Peat	Ceberg Peat
H-3	$1.2\cdot10^{-20}$	$3.2\cdot10^{-18}$	$1.3\cdot10^{-15}$	$3.6\cdot10^{-16}$	$2.6\cdot10^{-16}$
Be-10	$1.9\cdot10^{-18}$	$3.9\cdot10^{-16}$	$1.3\cdot10^{-14}$	$8.6\cdot10^{-13}$	$6.2\cdot10^{-13}$
C-14	$6.7\cdot10^{-18}$	$1.7\cdot10^{-15}$	$2.0\cdot10^{-17}$	$6.5\cdot10^{-15}$	$4.6\cdot10^{-15}$
Cl-36	$5.7\cdot10^{-19}$	$1.6\cdot10^{-16}$	$4.0\cdot10^{-13}$	$2.2\cdot10^{-11}$	$1.5\cdot10^{-11}$
Co-60	$3.3\cdot10^{-18}$	$5.1\cdot10^{-16}$	$5.0\cdot10^{-18}$	$2.8\cdot10^{-13}$	$2.7\cdot10^{-13}$
Ni-59	$9.0\cdot10^{-20}$	$2.2\cdot10^{-17}$	$1.1\cdot10^{-14}$	$2.7\cdot10^{-13}$	$1.9\cdot10^{-13}$
Ni-63	$2.1\cdot10^{-19}$	$5.2\cdot10^{-17}$	$7.6\cdot10^{-17}$	$1.6\cdot10^{-13}$	$1.4\cdot10^{-13}$
Se-79	$5.3\cdot10^{-17}$	$1.4\cdot10^{-14}$	$1.6\cdot10^{-12}$	$1.7\cdot10^{-9}$	$1.2\cdot10^{-9}$
Sr-90	$5.0\cdot10^{-18}$	$1.3\cdot10^{-15}$	$2.8\cdot10^{-14}$	$1.8\cdot10^{-11}$	$1.6\cdot10^{-11}$
Zr-93	$9.8\cdot10^{-19}$	$9.0\cdot10^{-17}$	$3.0\cdot10^{-15}$	$4.4\cdot10^{-13}$	$3.3\cdot10^{-13}$
Nb-94	$1.5\cdot10^{-18}$	$2.1\cdot10^{-16}$	$2.4\cdot10^{-13}$	$2.0\cdot10^{-12}$	$1.5\cdot10^{-12}$
Mo-93	$3.5\cdot10^{-19}$	$9.2\cdot10^{-17}$	$8.7\cdot10^{-13}$	$2.5\cdot10^{-12}$	$1.7\cdot10^{-12}$
Tc-99	$6.1\cdot10^{-19}$	$1.7\cdot10^{-17}$	$5.9\cdot10^{-14}$	$4.2\cdot10^{-13}$	$2.9\cdot10^{-13}$
I-129	$1.1\cdot10^{-16}$	$1.8\cdot10^{-14}$	$5.0\cdot10^{-11}$	$3.0\cdot10^{-11}$	$2.1\cdot10^{-11}$
Cs-135	$2.9\cdot10^{-18}$	$7.8\cdot10^{-16}$	$3.1\cdot10^{-13}$	$2.7\cdot10^{-12}$	$1.8\cdot10^{-12}$
Cs-137	$1.8\cdot10^{-17}$	$4.8\cdot10^{-15}$	$1.7\cdot10^{-16}$	$3.5\cdot10^{-12}$	$3.1\cdot10^{-12}$
Pb-210	$6.0\cdot10^{-16}$	$8.5\cdot10^{-14}$	$3.4\cdot10^{-15}$	$1.6\cdot10^{-11}$	$1.6\cdot10^{-11}$
Ra-226	$7.0\cdot10^{-17}$	$1.6\cdot10^{-14}$	$7.2\cdot10^{-12}$	$1.2\cdot10^{-9}$	$9.4\cdot10^{-10}$
Th-229	$4.0\cdot10^{-16}$	$1.3\cdot10^{-14}$	$4.6\cdot10^{-12}$	$7.0\cdot10^{-9}$	$6.5\cdot10^{-9}$
Th-230	$1.7\cdot10^{-16}$	$5.4\cdot10^{-15}$	$3.0\cdot10^{-12}$	$4.0\cdot10^{-9}$	$3.7\cdot10^{-9}$
Th-232	$1.9\cdot10^{-16}$	$5.9\cdot10^{-15}$	$3.4\cdot10^{-12}$	$4.4\cdot10^{-9}$	$4.1\cdot10^{-9}$
U-233	$1.1\cdot10^{-17}$	$2.6\cdot10^{-15}$	$3.7\cdot10^{-13}$	$6.1\cdot10^{-12}$	$4.3\cdot10^{-12}$
U-234	$1.1\cdot10^{-17}$	$2.5\cdot10^{-15}$	$3.6\cdot10^{-13}$	$5.9\cdot10^{-12}$	$4.1\cdot10^{-12}$
U-236	$1.0\cdot10^{-17}$	$2.4\cdot10^{-15}$	$3.4\cdot10^{-13}$	$5.5\cdot10^{-12}$	$3.8\cdot10^{-12}$
U-238	$9.7\cdot10^{-18}$	$2.3\cdot10^{-15}$	$3.1\cdot10^{-13}$	$5.1\cdot10^{-12}$	$3.5\cdot10^{-12}$
Np-237	$7.6\cdot10^{-18}$	$2.0\cdot10^{-15}$	$2.1\cdot10^{-12}$	$1.1\cdot10^{-10}$	$7.5\cdot10^{-11}$

8.6.2 Results

The radiation doses to which the calculated releases of radionuclides from SFL 3-5 to the biosphere can give rise have been calculated for the reference scenario on the three sites. The results for the nuclides that dominate the dose for releases from the different repository parts are presented in the following sections. Since we assume that all the releases from all three repository parts end up in the same ecosystem, the total dose given by the releases from SFL 3-5 is presented in conclusion.

The results are presented for times up to 10 million years after repository closure. In such a perspective, the future evolution of the environment is uncertain. Long-term climate changes include e.g. the formation of a continental ice sheet. Three ice ages are projected to occur during the next 100,000 years (Ahlbom *et al.*, 1991). The influence of this on SFL 3-5 is discussed in Section 9.2. In the results presented here, the uncertainty is indicated by shading the results from 100,000 years onward.

In accordance with SSI's (National Radiation Protection Institute) regulations regarding protection of human health and the environment in conjunction with the final disposal of spent nuclear fuel and nuclear waste (SSI, 1998), a final repository shall be designed so that the annual risk of deleterious effects after closure is no more than 10^{-6} for a representative individual in the group exposed to the greatest risk. The probability coefficients presented in the International Commission on Radiological Protection's (ICRP) Publication 60 (ICRP, 1991) are to be used for calculating the probability of deleterious effects. The effects to be taken into account according to SSI are cancer (lethal and non-lethal) and hereditary effects. The ICRP reports a probability coefficient of $7.3 \cdot 10^{-2}$ Sv^{-1} for this case. This probability of deleterious effects and an annual risk of 10^{-6} year^{-1} corresponds to an annual dose of 14 μSv for an individual, provided that the probability that this individual will be exposed to radionuclides from the repository is 100%. This dose has been included as a comparison level in the figures that present the results.

Radiation from the ground, outer space and radioactive substances occurring naturally in the human body gives each individual a dose in the order of 1 mSv/y. Radiation from other sources, e.g. radon in indoor air and medical radiation, results in a mean individual dose in Sweden of approximately 4 mSv/y. In the figures presenting the results of the calculations, the dose 1 mSv/y has been included as a background level.

SFL 3

In Aberg, the releases from SFL 3 will emerge into an area with coast and archipelago. Here there will be great dilution, so the doses resulting from the releases will be far below the comparison level. In the basic calculation case in the reference scenario, the dose is dominated by inorganic ^{14}C. The dose will reach a maximum after 10^4 years and reach $1 \cdot 10^{-9}$ Sv/y on release to the archipelago. If the release takes place instead at the coast, the dose is less than 10^{-10} Sv/y. In the case when radionuclide transport is affected by ISA, the dose is dominated by ^{210}Pb and the maximum dose is an order of magnitude higher. The release of ^{210}Pb comes only after a very long time, reaching a maximum after about 10^7 years.

In Beberg, the nuclides released from SFL 3 are discharged to an agricultural area. The dose is dominated by ^{93}Mo ($6\cdot10^{-7}$ Sv/y) and is obtained after more than 4,000 years, see Figure 8-10. After 20,000 years, ^{36}Cl makes the greatest contribution to the dose. If the release instead takes place in nearby peatland, ^{36}Cl gives the highest dose of $6\cdot10^{-6}$ Sv/y after 10,000 years. The composition of the groundwater in Beberg has a negligible influence on the aggregate dose.

In Ceberg, where peatland is the typical ecosystem, ^{36}Cl gives rise to the highest dose of $1\cdot10^{-6}$ Sv/y after approximately 30,000 years. ^{79}Se also makes a significant contribution to the aggregate dose, but only after 10^5 years.

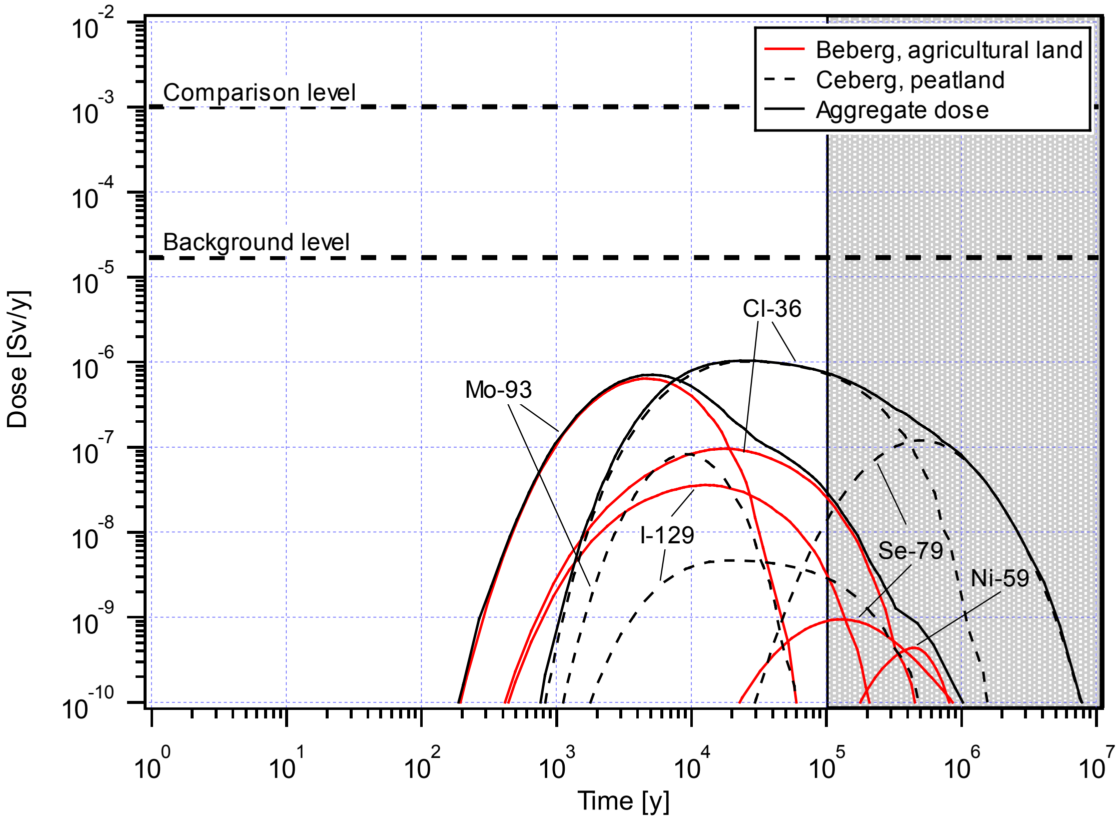

Figure 8-10 Dose from release of radionuclides from SFL 3 to agricultural land in Beberg and peatland in Ceberg. (For Beberg, the figure shows the calculation case with saline groundwater).

SFL 4

The calculated releases from SFL 4 are very small. On release to the archipelago area in Aberg, the dose is dominated by ^{90}Sr, which gives rise to a maximum dose of $3\cdot10^{-10}$ Sv/y after about 35 years. Other nuclides give a dose that does not exceed 10^{-10} Sv/y. Release to the coastal area in Aberg gives a maximum dose that is lower than 10^{-10} Sv/y. In Beberg, where the nuclides are released to agricultural land, ^{93}Mo dominates the dose, while ^{36}Cl dominates on the peatland in Ceberg. The dose from nuclides released in both Beberg and Ceberg is less than 10^{-10} Sv/y.

SFL 5

The calculated release from SFL 5 to the archipelago area in Aberg gives rise to two peaks in the dose curve. The first is from ^{3}H, which after 60 years gives the highest contribution of $1\cdot10^{-9}$ Sv/y. When the release of ^{3}H declines, the aggregate dose decreases, only to increase once again when the release of ^{93}Mo reaches the recipient. Between 5,000 and 500,000 years after repository closure – when ^{93}Mo, inorganic ^{14}C and ^{59}Ni successively dominate the release to archipelago – the aggregate dose is once again around 10^{-9} Sv/y. The maximum aggregate dose is $2\cdot10^{-9}$ Sv/y and is obtained after 20,000 years. As in the case of SFL 3 and SFL 4, the release to coast from SFL 5 gives an aggregate dose that never exceeds 10^{-10} Sv/y.

The nuclides that dominate the dose for release from SFL 5 to agricultural land in Beberg are ^{93}Mo and ^{36}Cl, see Figure 8-11. The maximum dose from ^{93}Mo is $2\cdot10^{-6}$ Sv/y after more than 7,000 years, while $7\cdot10^{-7}$ Sv/y is obtained from ^{36}Cl after 20,000 years. The salinity of the groundwater is not of crucial importance. If the release in Beberg takes place to peatland instead, the dose is dominated by ^{36}Cl and reaches $4\cdot10^{-5}$ Sv/y.

In Ceberg, peatland is the typical ecosystem and ^{36}Cl gives the maximum dose, $6\cdot10^{-6}$ Sv/y, after more than 20,000 years, see Figure 8-11. This is the highest dose obtained from SFL 3-5 for the typical ecosystems at the different sites. After times of up around a million years, ^{79}Se is projected to be the dominant dose.

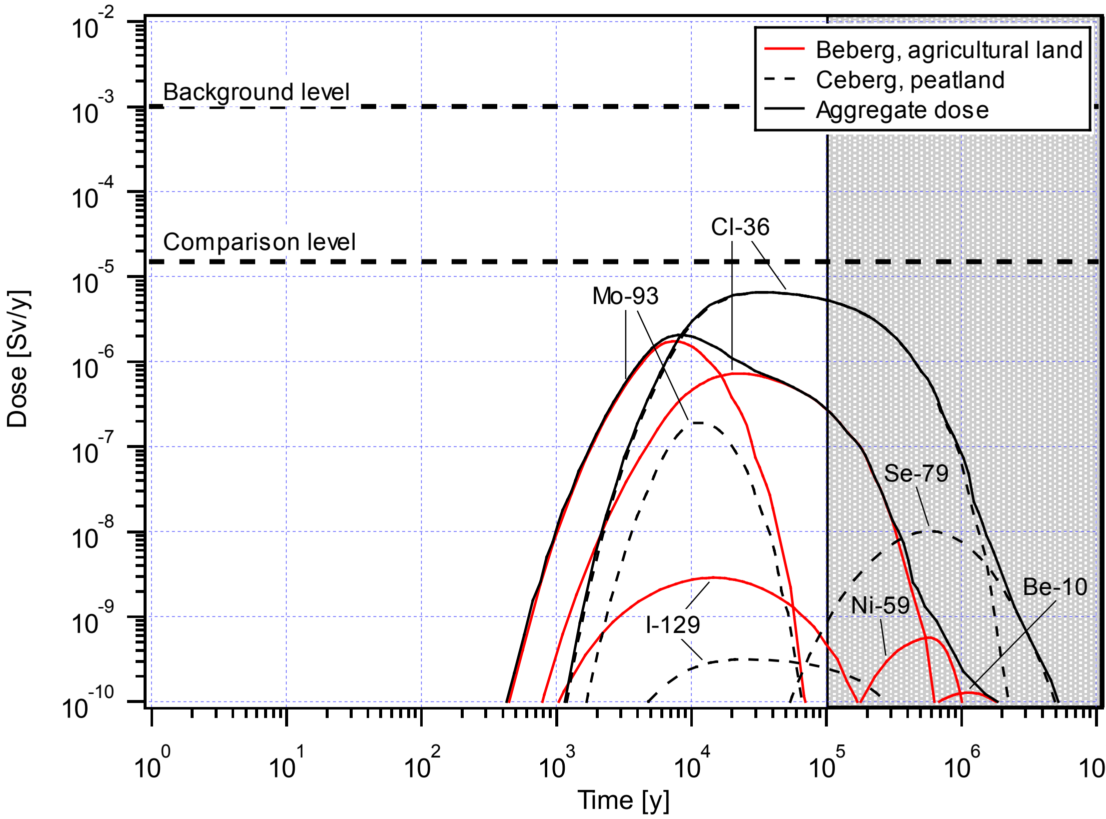

Figure 8-11 Dose from release of radionuclides from SFL 5 to agricultural land in Beberg and peatland in Ceberg. (For Beberg, the figure shows the calculation case with saline groundwater).

Total dose

So far, all results have been presented for one repository part at a time, but the releases from all three repository parts are expected to end up in the same ecosystem. The total dose is therefore of great interest, see Figures 8-12, 8-13 and 8-14. In the calculations, we have in this case assumed that the influence of ISA is negligible in SFL 3 and that CRUD has been washed away from the storage canisters in SFL 4.

It is only during a relatively short period after closure (up around 100 years) that SFL 4 makes any appreciable contribution to the total dose. After longer times, SFL 3 and SFL 5 alternate in making the biggest contribution to the total dose. Release to coast in Aberg gives a total dose that is 10–100 times lower than for release to archipelago. In all cases, the total dose lies below the comparison level.

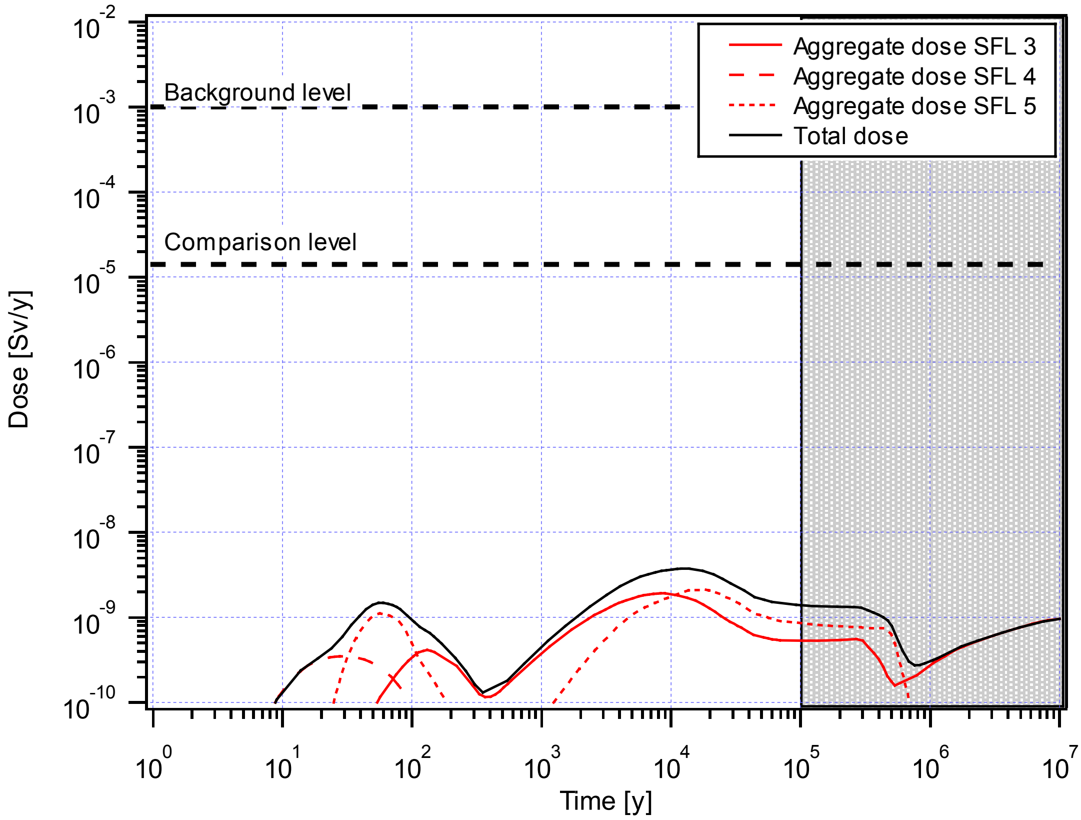

Figure 8-12 Total dose from release of radionuclides to archipelago in Aberg, with contributions from SFL 3, SFL 4 and SFL 5.

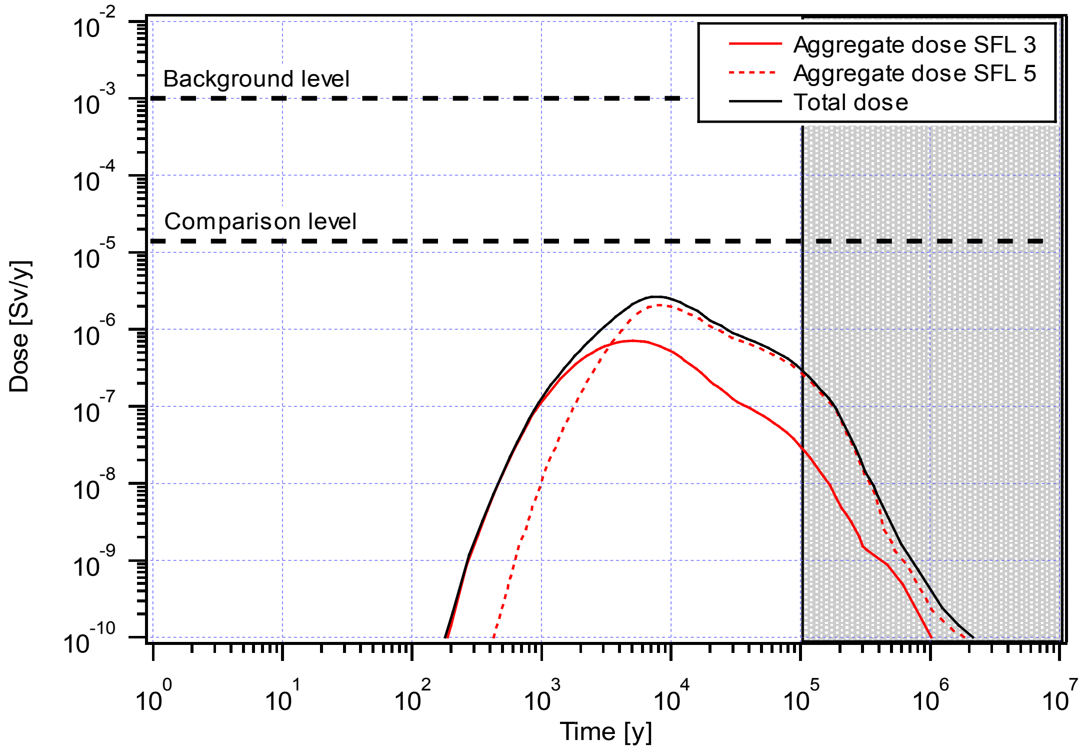

Figure 8-13 Total dose from release of radionuclides to agricultural land in Beberg, with contributions from SFL 3 and SFL 5.

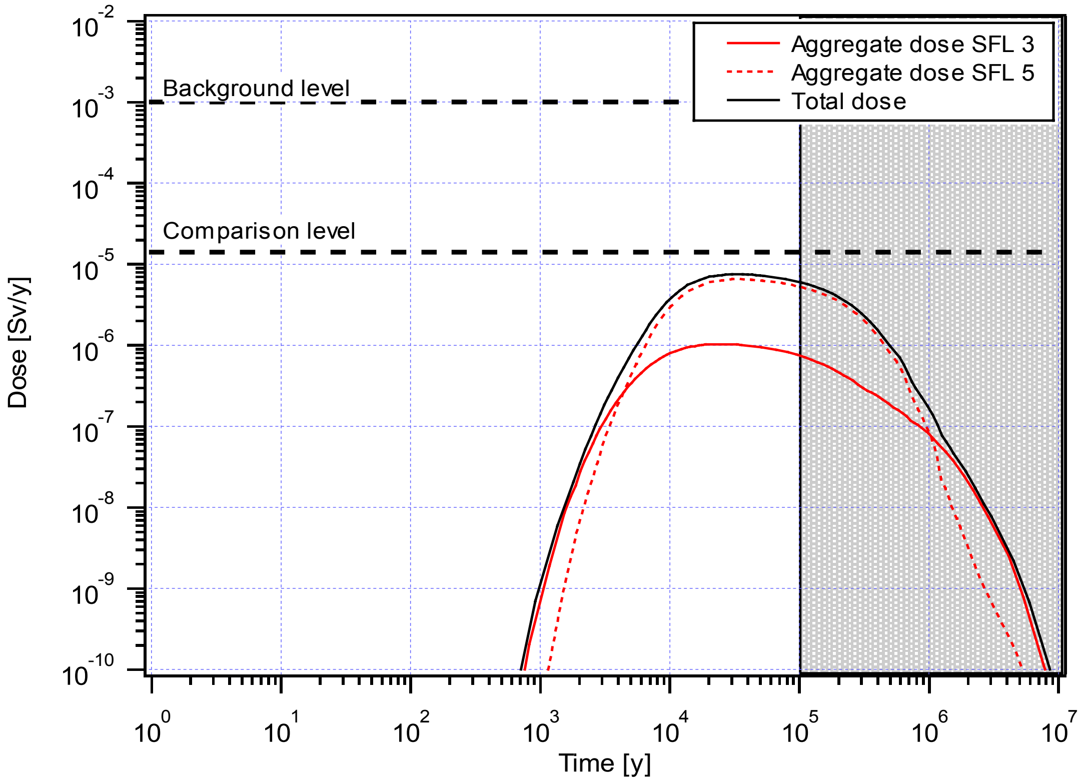

Figure 8-14 Total dose from release of radionuclides to peat in Ceberg, with contributions from SFL 3 and SFL 5.

8.7 Chemotoxic pollutants

The waste contains some substances that are hazardous to the environment but not particularly radioactive. These pollutants mainly include the metals lead, cadmium and beryllium, for example lead in special containers and cadmium or beryllium in scrap metal from research reactors (Chapter 2). The repository is supposed to protect man and the environment from these substances as well, and to verify this the metals Be, Cd and Pb are included in the safety assessment. The first step is to summarize the quantities of these metals in SFL 3, SFL 4 and SFL 5, see Table 8-17.

Table 8-17 Quantity of toxic metals in SFL 3-5 (kg).

Material	SFL 3	SFL 4	SFL 5	Total
cadmium	990		–	990
lead	19,965		108,592	128,557
beryllium	–		300	300

Dissolution and transport of toxic metals from the repository can naturally be calculated in the same way as for radionuclides, see Sections 8.3 and 8.4. It is also possible to estimate the consequences of a release in a similar manner by conservatively assuming that the entire calculated release goes to a typical ecosystem. The concentrations in the ecosystems are compared with concentrations measured in lakes, agricultural soils and peat, see Table 8-18.

If the ecosystem is open coast or archipelago, dilution occurs in relation to the water turnover rate. The water turnover rate in Aberg is $2.6 \cdot 10^7$ m^3/y out in the archipelago and $7.5 \cdot 10^9$ m^3/y at the open coast (Bergström *et al.*, 1999). The calculated cumulative release of chemotoxic metals to an archipelago in Aberg does not at any time give rise to concentrations that are higher than the assumed comparison levels.

For agricultural land and peatland, we assume roughly and at the same time conservatively that all material released from the geosphere enters the ecosystem and accumulates there. A concentration is finally obtained by dividing the accumulated quantity by the volume of the ecosystem module (Bergström *et al.*, 1999, Skagius *et al.*, 1999). The calculations show that the concentrations of metal in agricultural land or peatland are less than the comparison levels by a wide margin up to 10,000 years after closure. At this time, the concentrations are increasing and will exceed the comparison level between 100,000 and 10 million years after closure. By this time, however, several ice ages are expected to have come and gone, and it is unlikely that everything that has reached the ecosystem is still accumulated.

Table 8-18 Comparison levels for release of metals to different ecosystems.

Ecosystem	Be	Cd	Pb
Archipelago and Coast (µg/litre)	0.01 [a)]	0.01 [a)]	0.24 [a)]
Agriculture land (mg/kg dry solids)	0.6 [b)]	0.23 [c)]	17.1 [c)]
Peatland (mg/kg dry solids) [d)]	0.095	0.095 – 0.95	0.38 – 19

a) Mean concentrations in Swedish lakes and watercourses (Swedish EPA, 1999a

b) Mean concentrations in agricultural soil in USA (IPCS, 1990).

c) Mean concentrations in agricultural soil in Sweden (Swedish EPA, 1999b).

d) Typical value for peat used in energy production (Swedish EPA, 1983).

8.8 Release of radionuclides in gas phase

The waste in SFL 3-5 is not gaseous. However, the possibility that organic ^{14}C will eventually be transformed into e.g. methane cannot be ruled out. In this way, ^{14}C could accompany inactive gas from the repository, for example gas from corrosion of steel and aluminium or from degradation of organic materials. Organic ^{14}C is only contained in two types of waste in SFL 3 and the total inventory has been estimated at less than 10^5 Bq, see Table 8-1.

The collective dose from a pulse release of ^{14}C from one canister in a deep repository for spent nuclear fuel was calculated in the SR 95 study (SKB, 1995). The time for transport through the near and far field was neglected, i.e. the gas was assumed to be released immediately into the atmosphere. The release to the atmosphere was regarded as a release from reprocessing plants with a collective dose to the local and regional population of 0.4 mmanSv/GBq. With the same way of calculating, SFL 3 would at most give a collective dose of about 0.04 µmanSv locally and regionally. Since the collective doses to the local and regional population are distributed among a large number of individuals, the annual individual dose will be considerably lower.

8.9 References

Ahlbom K, Äikäs T, Ericsson L O, 1991. SKB/TVO Ice Age Scenario. SKB Technical Report TR 91-32, Svensk Kärnbränslehantering AB, Stockholm.

Andersson J, 1999. SR 97 – Data and data uncertainties. Compilation of data and evaluation of data uncertainties for radionuclide transport calculations. SKB Technical Report TR 99-09, Svensk Kärnbränslehantering AB, Stockholm.

Bergström U, Nordlinder S and Aggeryd I, 1999. Models for dose assessments. Modules for various biosphere types. SKB Technical Report TR 99-14, Svensk Kärnbränslehantering AB, Stockholm.

Carbol P, Engkvist I, 1997. Compilation of radionuclide sorption coefficients for performance assessment. SKB report R-97-13, Svensk Kärnbränslehantering AB, Stockholm.

Firestone R B, 1998. Table of Isotopes. 8th ed., 1998 Update, Ed. C.M. Baglin, John Wiley & Sons, Inc., N.Y. USA.

Holgersson S, Albinsson Y, Allard B, Borén H, Pavasars I, Engkvist I, 1999. The effects of isosaccharinic acid on Cs, Ni, Pm and Th sorption onto, and diffusion into cement. Part 2., Paper in preparation

IAEA, 1996. International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources. Safety Series No. 115, International Atomic Energy Agency, Vienna Austria.

ICRP, 1991. 1990, Recommendations of the International Commission on Radiological Protection. Annals of the ICRP, ICRP Publication 60, Pergamon Press, Oxford UK.

IPCS, 1990. Environmental Health Criteria 106; Beryllium. International Programme on Chemical Safety, World health Organisation, Geneva.

Naturvårdsverket (Swedish EPA), 1983. Miljöeffekter av ved och torvförbränning. PM 1708.

Naturvårdsverket (Swedish EPA), 1999a. Bedömningsgrunder för miljökvalitet - Sjöar och vattendrag. Rapport 4920, Naturvårdsverkets förlag, Stockholm.

Naturvårdsverket (Swedish EPA), 1999b. Bedömningsgrunder för miljökvalitet - Odlingslandskapet. Rapport 4916, Naturvårdsverkets förlag, Stockholm.

Nordlinder S, Bergström U, Mathiasson L, 1999. Ecosystem specific dose conversion factors for Aberg, Beberg and Ceberg. SKB Technical Report TR 99-15, Svensk Kärnbränslehantering AB, Stockholm.

Norman S, Kjellbert N, 1990. FARF31 – A far-field radionuclide migration code for use with the PROPER package. SKB Technical Report TR 90-01, Svensk Kärnbränslehantering AB, Stockholm.

Ohlsson Y, Neretnieks I, 1997. Diffusion data in granite. Recommended values. SKB Technical Report TR 97-20, Svensk Kärnbränslehantering AB, Stockholm.

Pettersson M, Moreno L, Skagius K, 1999. Analysis of radionuclide migration from SFL 3-5. SKB report R-99-14, Svensk Kärnbränslehantering AB, Stockholm.

Romero L, Moreno L, Neretnieks I, 1995. Fast multiple-path model to calculate radionuclide release from the near field of a repository. Nuclear Technology 112(1), pp 89-98.

Skagius K, Pettersson M, Wiborgh M, Albinsson Y, Holgersson S, 1999. Compilation of data for the analysis of radionuclide migration from SFL 3-5. SKB report R-99-13, Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1995. SR 95. Template for safety reports with descriptive example. Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1999. SR 97 – Main Report. Svensk Kärnbränslehantering AB, Stockholm.

SSI, 1998. Statens strålskyddsinsituts föreskrifter om skydd av människors hälsa och miljön vid slutligt omhändertagande av använt kärnbränsle och kärnavfall. SSI FS 1998:1, Statens strålskyddsinstitut.

9 Other scenarios

9.1 Introduction

In addition to the reference scenario described in Chapter 6, the following four scenarios are studied:

- Climate change
- Seismics/tectonics
- Future human actions
- Design and operation.

The scenarios studied are based on the choice of scenarios for the deep repository for spent fuel within the SR 97 study (SKB, 1999).

9.2 Climate change

The spans of time considered in the assessment of the environmental impact of a deep repository with waste containing long-lived radionuclides are so long that changes in climate can be expected to occur. In Scandinavia, long-term changes in climate have led to not only changes in the biosphere, but also the formation of permafrost and continental ice sheets. Changes similar to those we have seen in the past are expected to occur in the future as well.

The description of the Scandinavian climatic conditions in SR 97 (SKB, 1999) is based on three *climate-driven process domains*, which are:

- Temperate/boreal domain
- Permafrost domain
- Glacial domain.

The climate-driven process domains are supposed to provide a general picture of climate-driven conditions of importance for the evolution, performance and safety of the repository.

9.2.1 Climate scenario

During the Quaternary period, the past 2 million years, the earth's climate has had several global cold periods when continental ice sheets and glaciers have formed. The cold periods have been followed by briefer warm periods with a climate similar to that of today. The cold periods are termed *glacials* and the warm periods are termed *interglacials*. During the past 900,000 years or so, a pattern is repeated with approximately 100,000-year-long glacials, ending abruptly in a transition to a warm interglacial climate.

The long cold periods, the glacials, contain colder and warmer stages called *stadials* and *interstadials*, respectively. During glacial periods, continental ice sheets and permafrost have extended over Scandinavia. During the warmer interstadials, the extent of the ice sheet may have been limited to the Caledonian mountains. Southeast of the ice, conditions may have gradually changed from permafrost to a temperate/boreal domain. During the cold stages, or stadials, the ice may have covered large parts of Norway, Sweden and Finland, extending during the most extreme cold periods in over Russia in the east and down to Germany and Poland in the south.

SR 97's climate scenario is based on a number of relatively simple calculations of the climatic evolution during the next glacial and interglacial period. It describes climate change during the coming 130,000 years. Conditions vary considerably during a glacial. The fluctuations can be seen as a variation of the extent of the climate-driven process domains in time and space. The evolution on a given site can be described as a temporal series of climate-driven process domains (SKB, 1999).

A qualitative discussion of how the characteristic conditions within each climate-driven process domain influence SFL 3-5 and its environmental impact follows below. A comparison is made with the assessments in the reference scenario.

Temperate/boreal domain

The change within the temperate/boreal domain that is of the greatest importance for a repository is shoreline displacement. When the climate becomes colder, the sea level falls as water is bound in inland ice sheets all over the world. At the same time, the growing ice sheets cause downwarping of the earth's crust. In Sweden, crustal downwarping is generally greater than sea-level lowering. The sea level relative to a fixed point on the earth's crust – the *relative coastline* – rises during a glaciation.

The relative coastline rises as the ice sheet grows and the crust is depressed. Lowering of the relative coastline occurs during interglacial periods, when the previously ice-loaded and downwarped crust rebounds. The fastest lowering of the relative coastline occurs when the ice has just retreated from an area. Lowering of the relative coastline also occurs during the transition towards a colder climate due to the fact that the sea level falls.

When the relative coastline falls, the seabed becomes land. Groundwater flow gradients increase. Fresh precipitation water infiltrates the new land areas. When the relative coastline rises, land becomes sea once again. In the sea-covered areas, differences in water density can drive the groundwater flow. Salt water has a higher density than fresh water, and seawater can therefore infiltrate the bedrock.

Influence on SFL 3-5

The analyzed reference scenario applies for today's conditions, but does not take into account the changes in the geosphere and biosphere that take place during the period. These changes may cause the size and direction of the groundwater flow to change and lead to a different groundwater discharge area. Furthermore, the salinity of the groundwater may be affected. However, although the continuous change in these conditions has not been taken into account in the analysis of the reference scenario, the analysis shows the effects of different groundwater flows and primary recipients, as well as of saline or non-saline groundwater. These effects are discussed in Chapters 8 and 10.

Permafrost domain

Permafrost exists in areas where the ground temperature is below 0ºC for at least two consecutive years. Permafrost may be discontinuous or continuous. In areas with continuous permafrost, the ground is frozen everywhere, with the exception of major lakes and watercourses. In areas with discontinuous permafrost, sections with frozen ground are interspersed with unfrozen parts. In the transition towards temperate (boreal) conditions, the occurrence of permafrost becomes increasingly sporadic.

The climate is cold and relatively dry, with perhaps 50% of today's precipitation levels. Reduced precipitation and frozen ground result in very little groundwater recharge. The groundwater flow is limited to unfrozen sections of ground. The closely-spaced sections of frozen ground drive the groundwater flow to greater depths. Sections without permafrost are also frozen at the surface for most of the year. Most of the precipitation is cycled at the surface, so there is little water flow in the bedrock compared with present-day conditions. When the water in the rock fractures freezes, it expands and widens the fractures. Since the deformation is not fully elastic, the widening is partially permanent. The low temperatures lower reaction and dissolution rates. At the same time, the solubility of carbon dioxide increases, which means that the solubility of calcite increases. Water is frozen out, which means that salinity will increase.

Of the three repository sites, it is only in Ceberg that permafrost is expected to occur during the next 130,000 years. The permafrost is discontinuous and is not expected to reach any great depths. When the coastline rises and all large areas are covered by the sea, the permafrost domain will no longer exist.

Influence on SFL 3-5

Permafrost entails that the water in the ground is frozen all year long. This prevents migration and helps to seal the repository. If, on the other hand, the frost should go so deep that the water freezes in the repository itself, this could at worst lead to damage to the barriers in the near field caused by bursting. This means that the water flow through the repository barriers could increase in a later phase. If large quantities of water freeze, the salinity of the water that has not yet frozen may increase (Karlsson *et al.*, 1999). The effect of different water flows and of different salinities in the water is analyzed in the reference scenario, see Chapters 8 and 10.

Glacial domain

A glacial domain prevails in the presence of a continental ice sheet. Under the central portions of the ice sheet – the *ice divide zone* – the ground is frozen. The permafrost depth may be up around 400 m. Nearer the front of the ice sheet the pressure melting point is reached and a zone is formed where melting occurs in the interface between ground and ice, the *melting zone*.

In an analysis of the groundwater flow under glaciers, Boulton *et al.* (1999) suggest that the meltwater is mainly transported towards the ice marginal through ice tunnels in the ice/bedrock interface. The meltwater is transported in to the ice tunnels by groundwater flow. Water pressure and flow when the ice marginal lies in the sea are shown in Figure 9-1. The gradients for groundwater flows are high at the ice marginal and in towards the ice tunnels. Groundwater flow takes place both in towards the ice tunnels and out towards the ice marginal. The glacial meltwater is oxygen-rich. In the ice divide zone the ground is frozen and no groundwater flows occur.

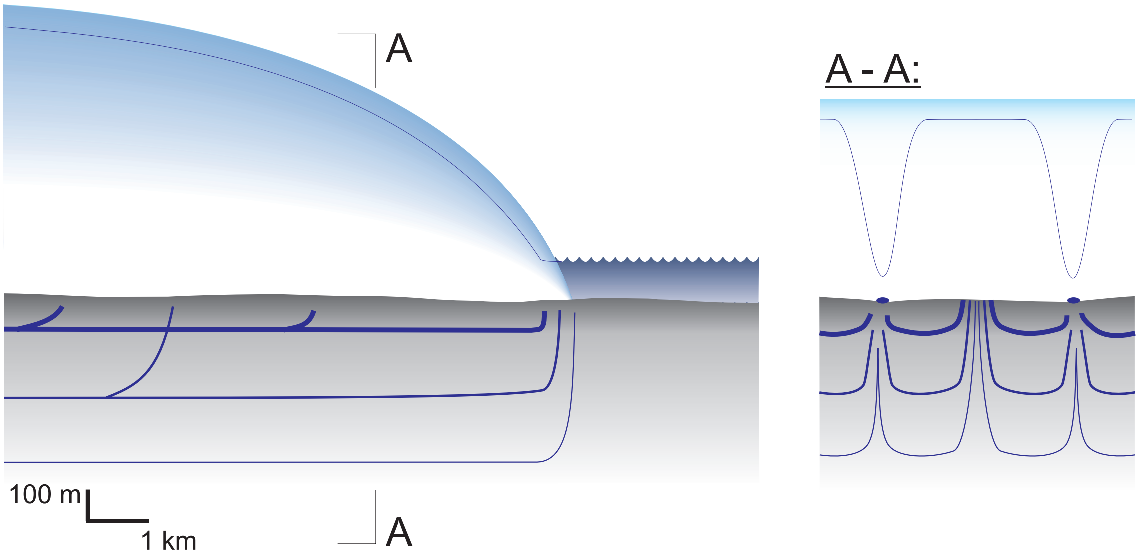

Figure 9-1 Groundwater flow in the melting zone. The ice marginal is in the sea.

When a continental ice sheet moves over an area, the prevailing rock stresses are affected. In the ice divide zone where the ice cover is thickest, the stresses are great. The bedrock is compressed and fractures are squeezed together. The ice in the melting zone is less thick and the water pressure is locally high. The high water pressures can lead to widening of fractures, resulting in increased water permeability.

Influence on SFL 3-5

A continental ice sheet on the surface exerts a violent impact on the landscape, but this is of less importance for the repository, which is located 300 m or deeper down in the rock. If the ice cover grows very thick, however, the rock stresses at repository depth increase and this can affect and possibly damage the rock caverns and tunnels in the repository.

After an ice age, as the ice melts at the margin of the ice sheet, a situation might arise where the gradient, and thereby also the flow, of groundwater increases (Ahlbom *et al.*, 1991). The effect of different groundwater flows is included in the analysis of the reference scenario, see Chapters 8 and 10.

The meltwater has a low concentration of dissolved salts and has a greater chance of preserving its dissolved atmospheric oxygen during the infiltration due to the ice and the cold climate (Karlsson *et al.*, 1999). Nevertheless, it is not likely that this will appreciably affect the repository in any way. The melting period at any given place is limited, and the repository has a great inherent capacity to absorb and react with oxygen due to its content of metals and organic substances. Furthermore, relatively few nuclides are adversely affected by oxidation.

9.3 Seismics/tectonics

An investigation regarding earthquakes and possible displacements in different fracture systems has been carried out in the SR 97 study (La Pointe *et al.,* 1997). Two main scenarios for the influence of an earthquake on the repository have been studied:

- Earthquake along a fault intersecting the repository
- Earthquake along a fault that does not intersect the repository, but that induces displacements in secondary fractures in the repository area.

The information in a global database shows that for a quake with a magnitude of 5.5 on the Richter scale, which is comparable with quakes measured in Sweden during a 1,000-year period, shear movements of about 0.05 metre are obtained in a “normal” fault.

Displacements in small rock structures in the repository area caused by powerful earthquakes in steeply dipping faults at different distances from the repository have been calculated for Äspö rock. The results show that earthquakes with a magnitude of 6.0–6.5 give rise to displacements of 0.01–0.02 m at a distance of 50 metres between fault and repository area. An earthquake at a greater distance from the repository, 2 km, and with a magnitude of 6.1 gives a maximum displacement of about 1 mm in minor rock structures in the repository area.

9.3.1 Influence of earthquake on SFL 3 and SFL 5

The concrete enclosures are founded on a compacted fill of crushed rock with a thickness of at least 0.5 metre and are surrounded by a gravel backfill that is more than 1 metre thick. The concrete structure has a total length of about 114 metres and is divided into three sections, each 38 metres long. The sections are separated by expansion joints.

A local minor displacement in the bedrock is absorbed in the foundation material and by concrete deformations. It is believed that an isolated and instantaneous local displacement in the bedrock of the order of 10 mm will not lead to harmful cracking in the concrete structure. If the displacement is instead divided up into several smaller displacement sequences that occur at long time intervals (hundreds of years), it is probable that the aggregate displacement without damage to the concrete can be two to three times greater due to plasticization and creep in the concrete and the foundation material.

It is possible to adopt some measures to increase the deformation resistance of the concrete structure, for example:

- more expansion joints

- locating the expansions joints at fractures in the rock that can be expected to be movement-prone in the event of an earthquake
- thicker layer of crushed rock/gravel backfill
- adjusted particle size distribution of the crushed rock/gravel backfill.

9.3.2 Influence of earthquake on SFL 4

In SFL 4 there is a thin gravel layer underneath the concrete base slab. Furthermore there is a 0.4 metre thick gravel backfill between the waste packages and rock walls, plus a larger volume on top of the waste packages. In the calculations of the radionuclide release from SFL 4 in the reference scenario, the waste packages are not considered to present any obstacle to the nuclides, so a fracture displacement does not entail any changes in the assessment of environmental impact.

9.4 Future human actions

Examples of future human actions that might affect the performance and thereby the safety of a deep repository for spent fuel have been identified within the SR 97 study, see Table 9-1 (Morén *et al.*, 1998). Thermal impact could arise if a heat store is built in the future or an attempt is made to extract geothermal energy. Hydrological impact could occur if water installations are built, for example for hydropower and irrigation. Drilling and construction of rock caverns, tunnels and shafts in conjunction with the repository could also lead to mechanical impact on the repository. Injection of hazardous waste and acidification of air and soil are examples of chemical impact. For SFL 3-5, we have chosen to shed light on the influence of future human actions by calculating future environmental impacts from drilled wells in the vicinity of the repository.

Table 9-1 Human actions that might influence a deep repository divided into the categories "Thermal", "Hydrological", "Mechanical" and "Chemical" impact (Morén *et al.*, 1998).

Category	Action
Thermal impact	Build heat store Build heat pump installation Extract geothermal energy (geothermics) Build facility that generates heat/cold on the surface of the repository
Hydrological impact	Sink well Build dam Change the course, size and interconnections of bodies of surface water (rivers, streams, lakes, sea) Build hydropower installation Build drainage installation Build infiltration installation Build irrigation installation Change conditions for groundwater recharge by changed land use
Mechanical impact	Drill in the rock Build rock caverns, tunnels, shafts, etc. Excavate open-pit mine Establish landfill Bomb or blast on the surface of the repository
Chemical impact	Dispose of hazardous waste in the rock Establish landfill Acidify air and soil Sterilize soil Cause accident resulting in chemical pollution

9.4.1 Wells

According to the regional hydrology models, no releases take place from SFL 3-5 to any of the wells that have been sunk today in the release area in Aberg, Beberg or Ceberg. However, it is conceivable that a well could be sunk on such a site in the future. The release of radionuclides and toxic metals to a well that is assumed to lie in the discharge area is discussed below (Petterson *et al.*, 1999). Dose impact from well during the first hundred years has been shaded, since it has been assumed that the repository site is protected by institutional controls and/or that the existence of the repository is known for these times. It is considered reasonable that it will take at least three generations, or of the order of 100 years, before knowledge of a repository can be said to be lost (Morén *et al.*, 1998).

Site-specific average wells

Site-specific average wells have been defined as wells with the same average capacity as the wells that exist today within a given release area. This gives us three average wells with capacities of 300 l/h (Aberg), 1,000 l/h (Beberg) and 500 l/h (Ceberg).

Release of radionuclides to well

The model that underlies the ecosystem-specific dose conversion factors, EDFs, calculated for the three average wells in Table 9-2 is presented in Chapter 5. Besides water capacity, the site-specific EDFs calculated for the three average wells also take into account precipitation conditions on the site and associated differences in water use.

Table 9-2 Mean values of ecosystem-specific dose conversion factors (Sv/Bq) for average wells (Nordlinder *et al.*, 1999)

Radio-nuclide	Aberg Average well (300 l/h)	Beberg Average well (1,000 l/h)	Ceberg Average well (500 l/h)
H-3	$9.4\cdot10^{-15}$	$2.7\cdot10^{-15}$	$5.2\cdot10^{-15}$
Be-10	$4.6\cdot10^{-13}$	$1.3\cdot10^{-13}$	$2.6\cdot10^{-13}$
C-14	$2.4\cdot10^{-13}$	$7.0\cdot10^{-14}$	$1.4\cdot10^{-13}$
Cl-36	$7.3\cdot10^{-13}$	$2.1\cdot10^{-13}$	$3.7\cdot10^{-13}$
Co-60	$1.1\cdot10^{-12}$	$3.3\cdot10^{-13}$	$6.6\cdot10^{-13}$
Ni-59	$5.9\cdot10^{-14}$	$1.8\cdot10^{-14}$	$3.0\cdot10^{-14}$
Ni-63	$6.2\cdot10^{-14}$	$1.9\cdot10^{-14}$	$3.7\cdot10^{-14}$
Se-79	$2.7\cdot10^{-12}$	$7.7\cdot10^{-13}$	$1.3\cdot10^{-12}$
Sr-90	$1.3\cdot10^{-11}$	$3.7\cdot10^{-12}$	$7.3\cdot10^{-12}$
Zr-93	$3.7\cdot10^{-13}$	$1.1\cdot10^{-13}$	$2.2\cdot10^{-13}$
Nb-94	$3.4\cdot10^{-12}$	$9.8\cdot10^{-13}$	$1.7\cdot10^{-12}$
Mo-93	$2.5\cdot10^{-12}$	$7.7\cdot10^{-13}$	$1.2\cdot10^{-12}$
Tc-99	$5.5\cdot10^{-13}$	$1.6\cdot10^{-13}$	$2.8\cdot10^{-13}$
I-129	$9.2\cdot10^{-11}$	$2.7\cdot10^{-11}$	$4.6\cdot10^{-11}$
Cs-135	$1.9\cdot10^{-12}$	$5.7\cdot10^{-13}$	$1.0\cdot10^{-12}$
Cs-137	$5.6\cdot10^{-12}$	$1.7\cdot10^{-12}$	$3.4\cdot10^{-12}$
Pb-210	$1.9\cdot10^{-10}$	$5.6\cdot10^{-11}$	$1.1\cdot10^{-10}$
Ra-226	$1.2\cdot10^{-10}$	$3.4\cdot10^{-11}$	$6.5\cdot10^{-11}$
Th-229	$4.2\cdot10^{-10}$	$1.3\cdot10^{-10}$	$2.4\cdot10^{-10}$
Th-230	$2.1\cdot10^{-10}$	$6.2\cdot10^{-11}$	$1.2\cdot10^{-10}$
Th-232	$2.3\cdot10^{-10}$	$6.8\cdot10^{-11}$	$1.3\cdot10^{-10}$
U-233	$1.9\cdot10^{-11}$	$5.5\cdot10^{-12}$	$1.1\cdot10^{-11}$
U-234	$1.8\cdot10^{-11}$	$5.3\cdot10^{-12}$	$1.0\cdot10^{-11}$
U-236	$1.7\cdot10^{-11}$	$5.0\cdot10^{-12}$	$9.7\cdot10^{-12}$
U-238	$1.6\cdot10^{-11}$	$4.8\cdot10^{-12}$	$9.2\cdot10^{-12}$
Np-237	$4.7\cdot10^{-11}$	$1.3\cdot10^{-11}$	$2.5\cdot10^{-11}$

The consequences of radionuclide releases to the site-specific average wells that have been defined are illustrated on the following pages. It is assumed that nuclides reach the well at the same rate (Bq/y) as they leave the far field, and without any delay. The results for SFL 3 and SFL 4 are based on the assumption that ISA does not have any influence on the near-field transport of radionuclides from SFL 3 and that all CRUD has been washed off before deposition in SFL 4. Nuclides that dominate the dose and the dose obtained from the release of all nuclides in the well (aggregate dose) are given in the figures. The comparison level 14 µSv/y and the background level 1 mSv/y are also included in the figures, see Section 8.6.2.

Aberg

The releases of dose-dominant nuclides from SFL 3-5 to the assumed average well in Aberg are given in Figures 9-2 to 9-4.

Release of ^{3}H and ^{90}Sr from SFL 3 to the average well in Aberg gives a maximum dose of 0.2 µSv/y and 4 µSv/y, respectively (Figure 9-2). Most of the release reaches the well during the institutional control period. After longer times, the dose is dominated by ^{93}Mo and ^{59}Ni. The maximum dose is 6 µSv/y and comes from ^{93}Mo after approximately 2,000 to 3,000 years. In the case when radionuclide transport is influenced by ISA, the dose is dominated by ^{210}Pb and the maximum dose is an order of magnitude higher. The release of ^{210}Pb comes only after a very long time, with a calculated maximum after about 10^{7} years.

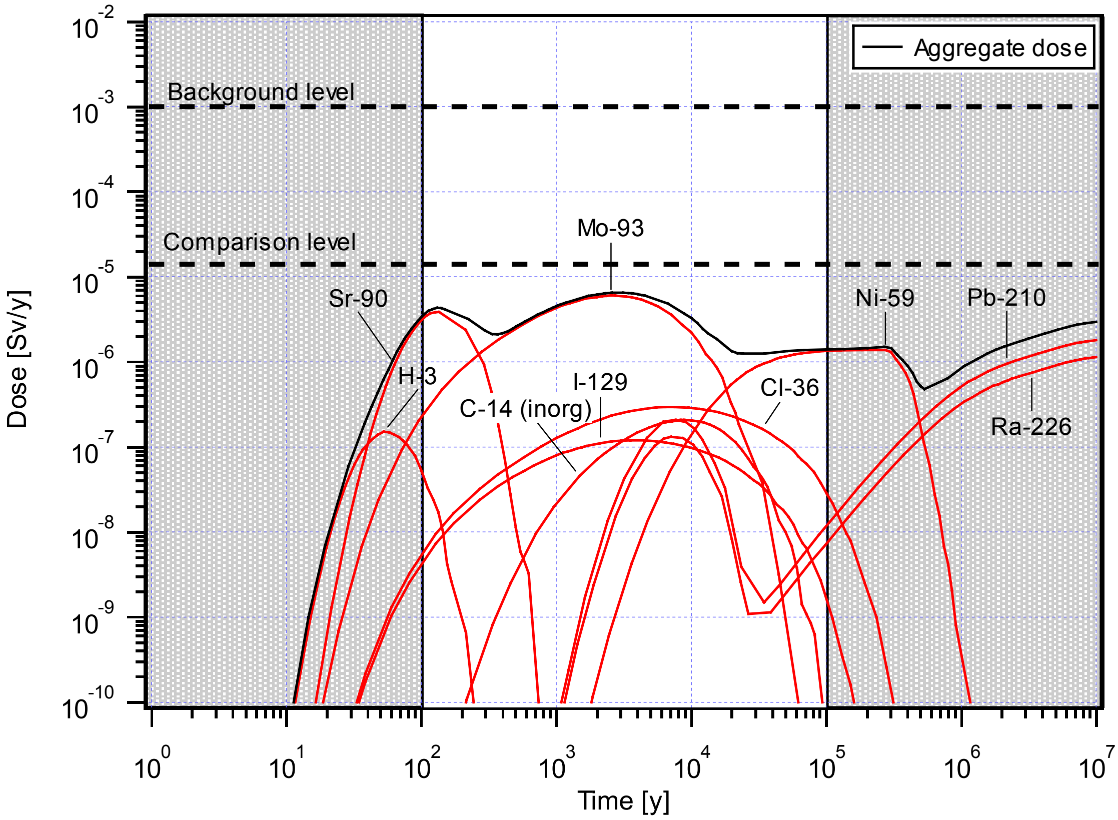

Figure 9-2 Dose for radionuclide release from SFL 3 to average well in Aberg (no influence of ISA on solubilities in the near field).

The calculation of radionuclide migration from SFL 4 is based on the assumption that all CRUD on the storage canisters has been washed off before deposition. The content of radionuclides in SFL 4 will thereby be low, which means that the release of radionuclides from the far field declines relatively rapidly with time, see Figure 8-8.

This is also reflected in how the dose varies with time (Figure 9-3). The release of ^{90}Sr, which gives the highest dose (3 μSv/y), mainly reaches the recipient during the institutional control period. After this period the dose has declined and is far below the comparison level.

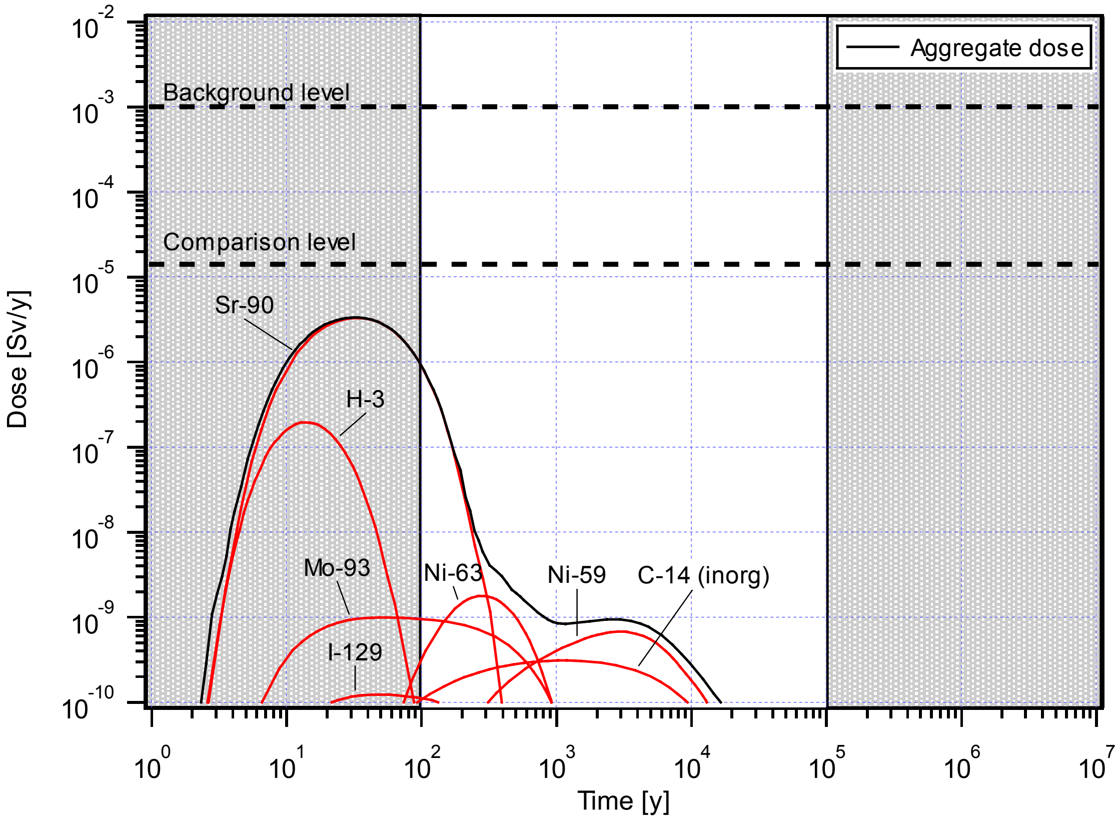

Figure 9-3 Dose for radionuclide release from SFL 4 (without CRUD) to average well in Aberg.

During the institutional control period, the release from SFL 5 to the average well in Aberg is dominated by ^{3}H (Figure 9-4). After that the release is dominated primarily by ^{93}Mo, but ^{90}Sr, ^{36}Cl, ^{59}Ni and ^{93}Zr also make substantial contributions to the dose at different times. The maximum dose to which the release of ^{93}Mo gives rise is the highest obtained in the basic calculation case within the reference scenario, 18 μSv/y.

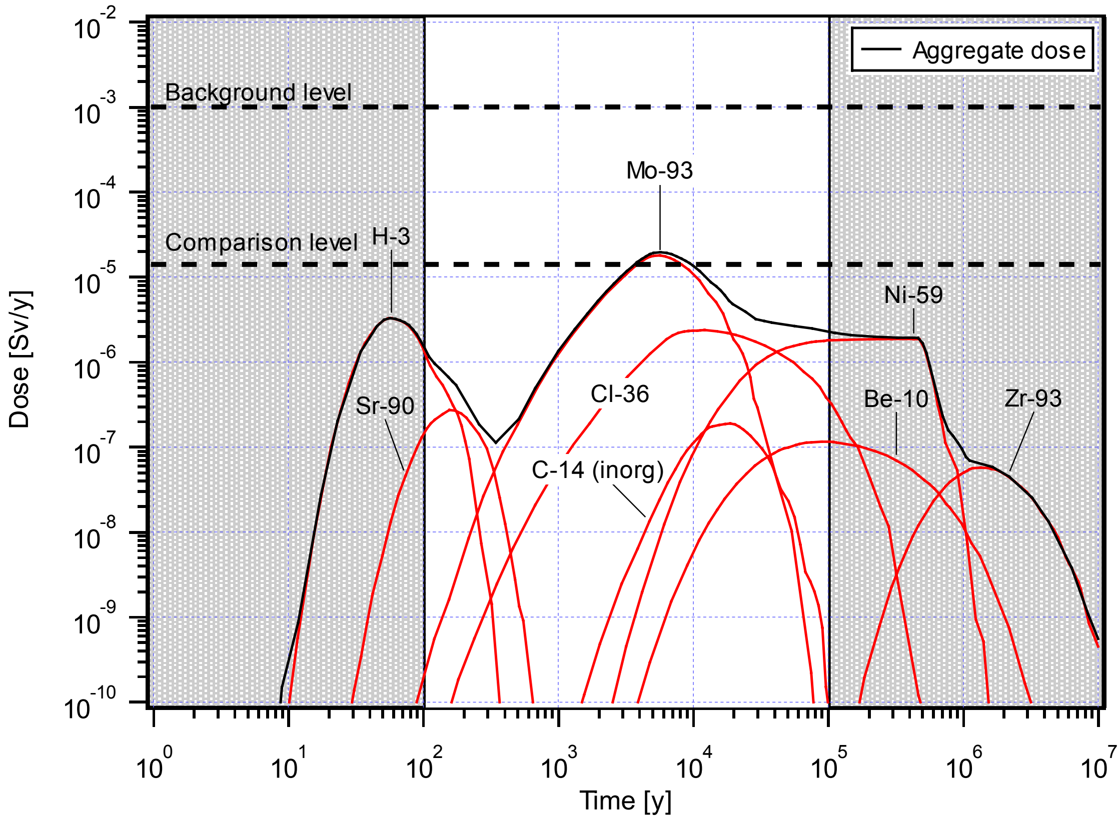

Figure 9-4 Dose for radionuclide release from SFL 5 to average well in Aberg.

Beberg

The releases of dose-dominant nuclides from SFL 3 and SFL 5 to the assumed average well in Beberg are presented in Figures 9-5 and 9-6. The results are based on a saline groundwater, but the composition of the groundwater in Beberg has a negligible influence on the dose. The release of short-lived nuclides, such as ^{3}H and ^{90}Sr, from the far field in Beberg is low, which means that the releases to well give very low doses during the institutional control period.

The dose from releases of all nuclides from SFL 3 rises up to about 4,000 years after closure, after which it declines again. Dominant nuclides are ^{93}Mo and ^{36}Cl. The former gives rise to the maximum dose, 0.6 µSv/y.

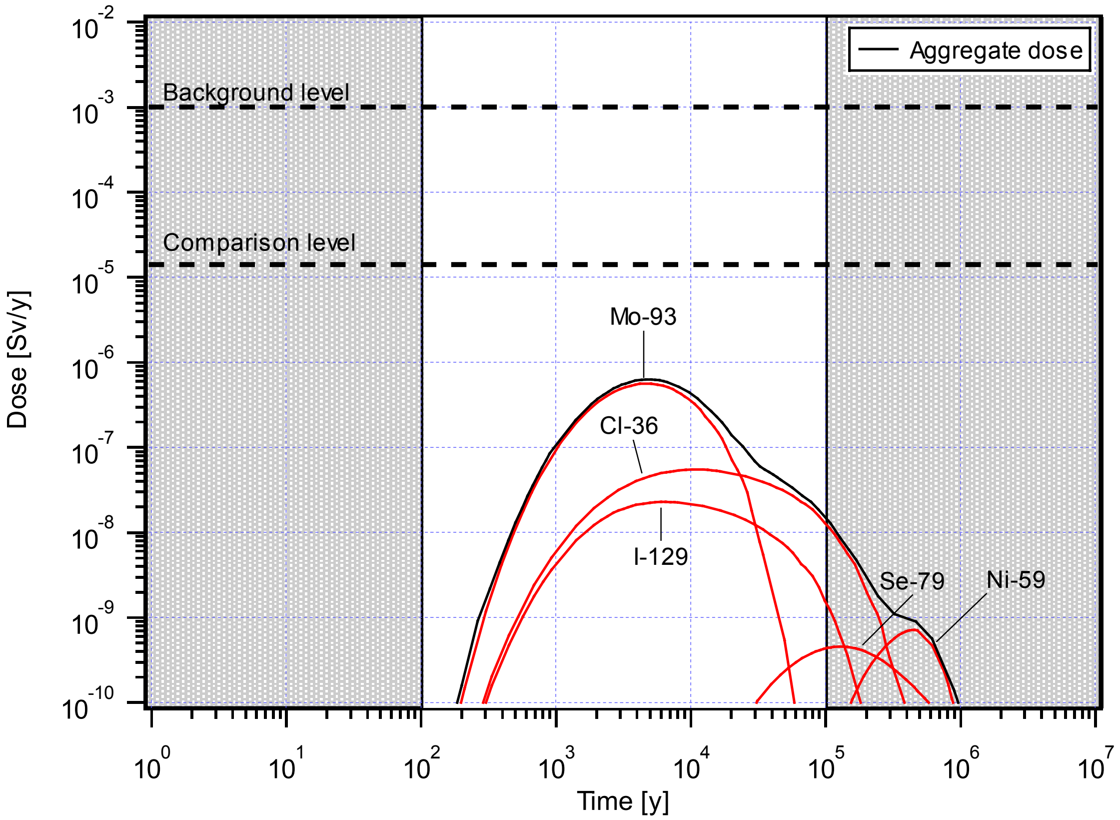

Figure 9-5 Dose for radionuclide release from SFL 3 to average well in Beberg. (The figure shows the calculation case with saline groundwater and no influence of ISA on solubilities in the near field).

The releases to the average well from SFL 4 in Beberg are so low that the dose does not exceed 10^{-10} Sv/y.

More or less the same nuclides dominate the dose from the release to the average well in Beberg from SFL 5 as from SFL 3, i.e. ^{93}Mo and ^{36}Cl. The maximum dose is 2 µSv/y, which is obtained after about 7,000 years.

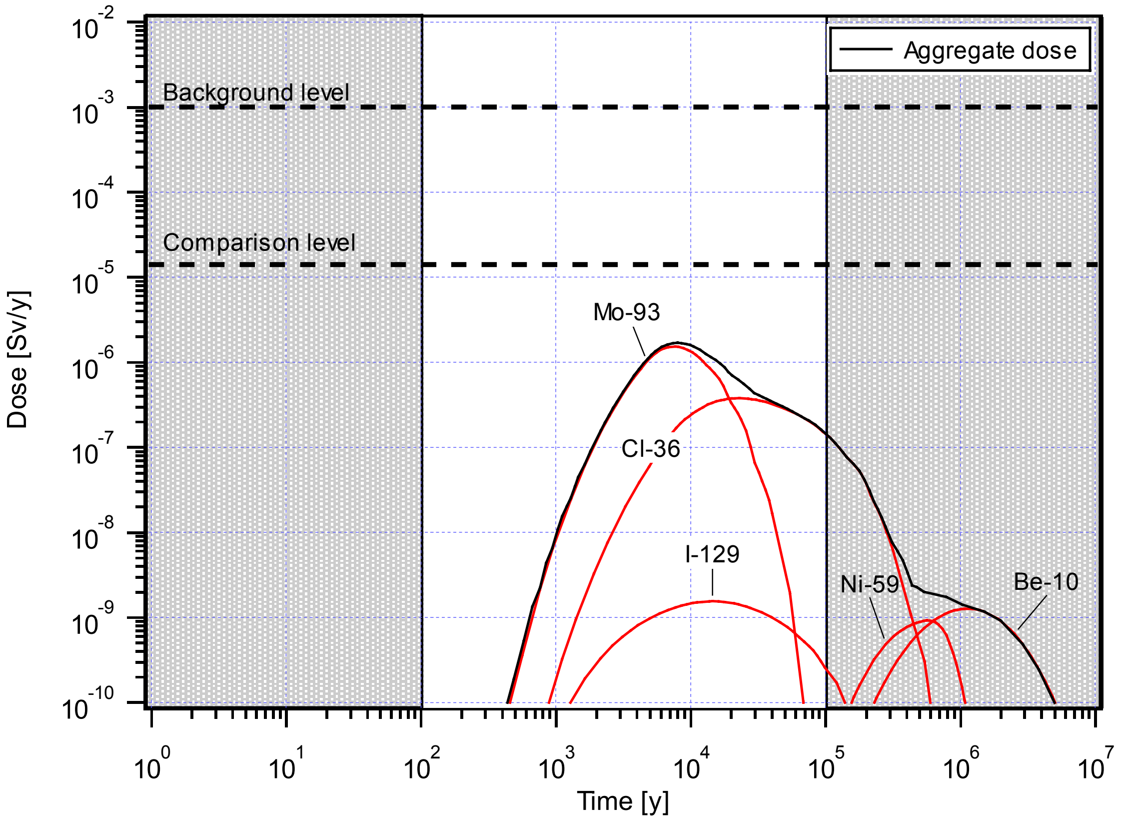

Figure 9-6 Dose for radionuclide release from SFL 5 to average well in Beberg. (The figure shows the calculation case with saline groundwater).

Ceberg

The releases of dose-dominant nuclides from SFL 3 and SFL 5 to the assumed average well in Ceberg are presented in Figures 9-7 and 9-8, respectively. In keeping with the results for Beberg, release to the average well in Ceberg gives very low doses during the institutional control period. The maximum dose is not obtained until approximately 10,000 years after closure. The nuclides that dominate the dose are in both cases ^{93}Mo and ^{36}Cl. For release from SFL 3, ^{93}Mo gives the highest dose (0.06 µSv/y) after approximately 9,000 years. The maximum dose for release from SFL 5 (0.2 µSv/y) is obtained about 30,000 years after closure and comes from ^{36}Cl.

The releases from SFL 4 in Ceberg are so low that the dose for release to the average well does not exceed 10^{-10} Sv/y.

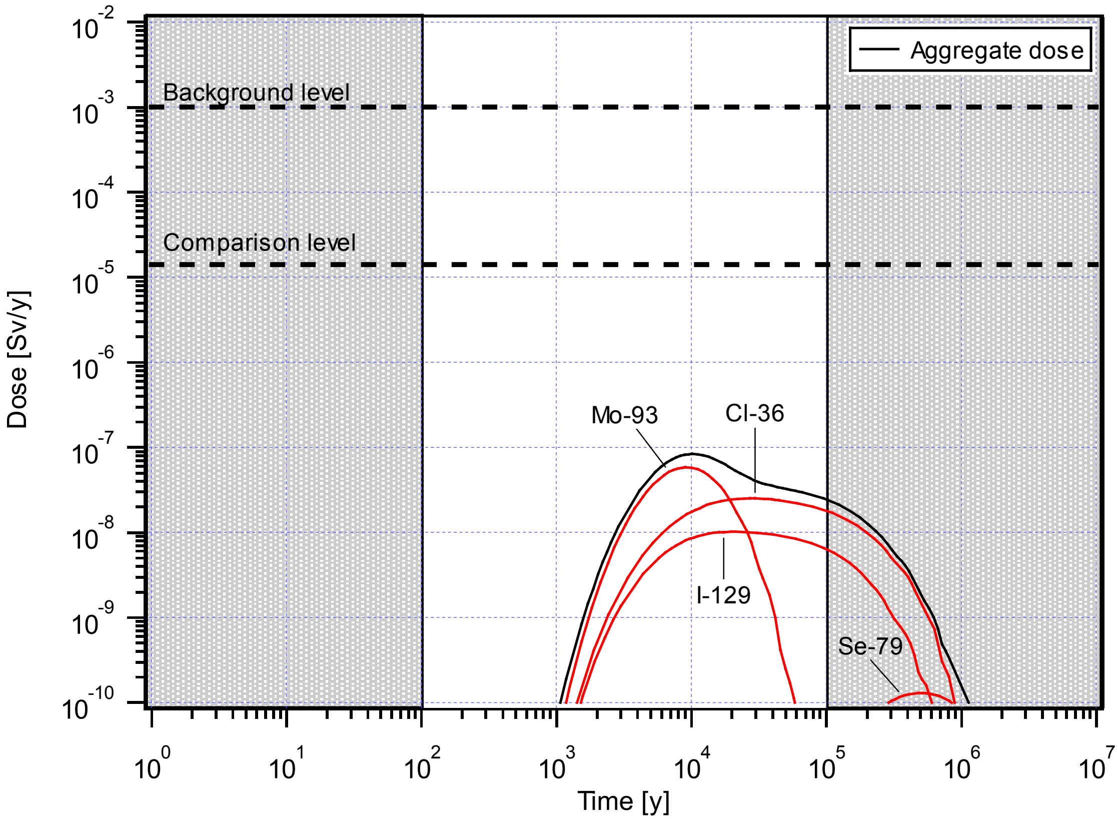

Figure 9-7 Dose for radionuclide release from SFL 3 to average well in Ceberg (no influence of ISA on solubilities in the near field).

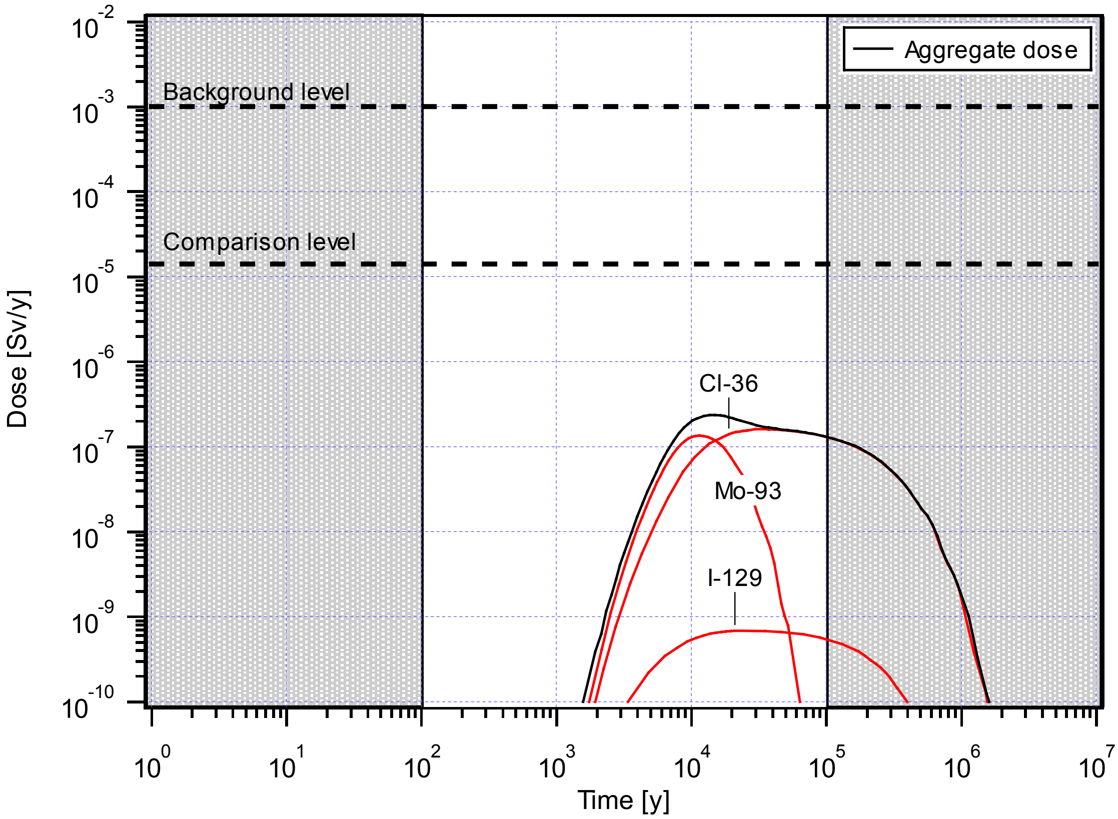

Figure 9-8 Dose for radionuclide release from SFL 5 to average well in Ceberg.

Total dose

When the releases from all three repository parts are expected to end up in the same recipient, the aggregate dose from all repository parts is of the utmost interest. The dose obtained for the assumed average wells is shown in Figures 9-9, 9-10 and 9-11. The calculations have been based on the assumption that the influence of ISA on solubilities in the near field is negligible in SFL 3 and that CRUD has been washed off of the storage canisters in SFL 4.

It is primarily SFL 3 and SFL 5 that make substantial contributions to the total dose. Only during the first 100 years after closure does SFL 4 make any substantial contribution. The highest values of the dose from all repository parts are 25 µSv/y (Aberg), 2 µSv/y (Beberg) and 0.3 µSv/y (Ceberg). The influence of ISA on radionuclide transport in SFL 3 also has an effect on the total dose, although after very long times. In Aberg, the total dose increases for times of around 10^6 years and longer by an order of magnitude compared with the results presented in Figure 9-9. The effect is smaller in Beberg and Ceberg and furthermore occurs even later.

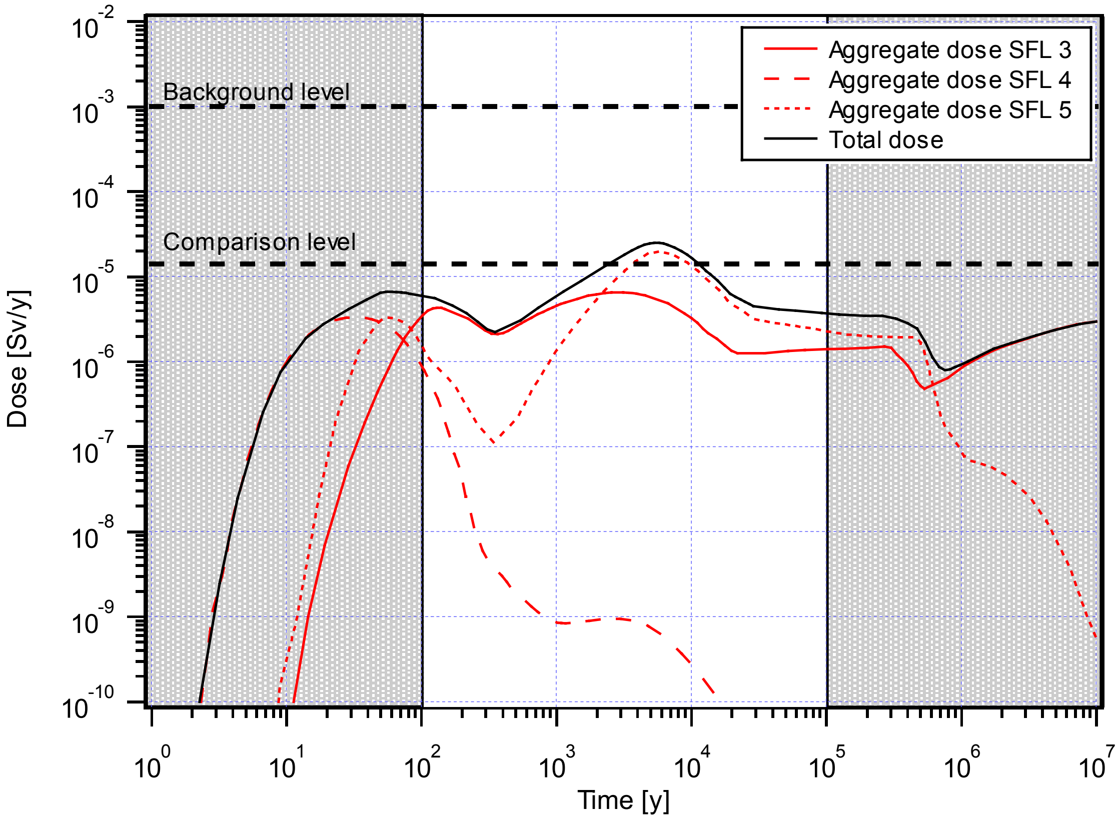

Figure 9-9 Total dose for radionuclide release to average well in Aberg.

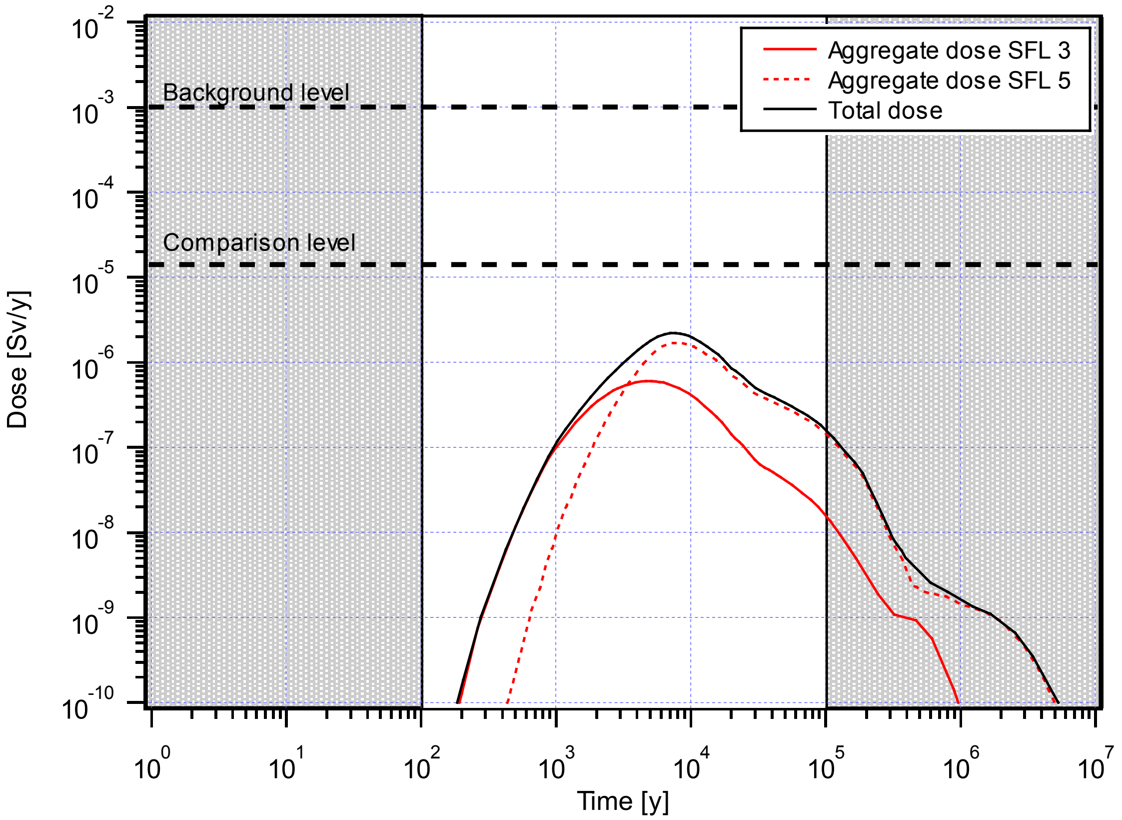

Figure 9-10 Total dose for radionuclide release to average well in Beberg. (The figure shows the calculation case with saline groundwater).

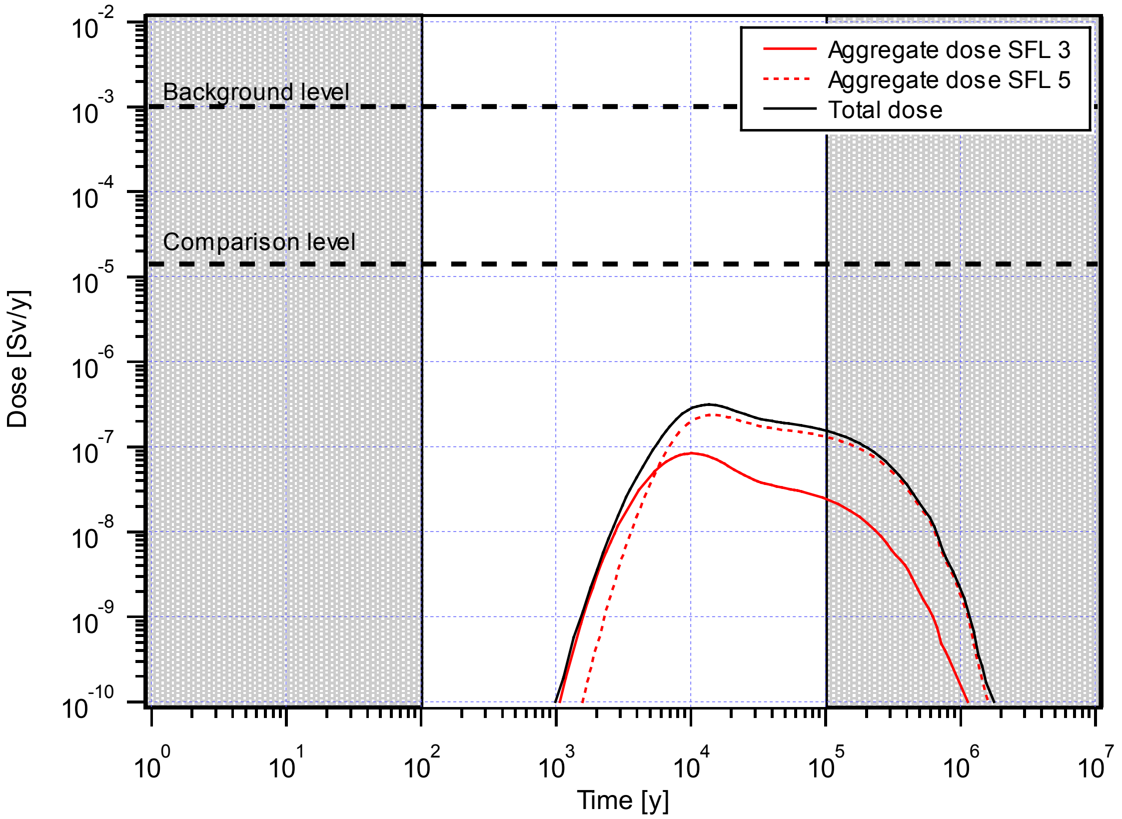

Figure 9-11 Total dose for radionuclide release to average well in Ceberg.

Release of chemotoxic pollutants to well

The importance of release of the toxic metals lead, beryllium and cadmium to a well has been investigated. The results are based on the same assumptions as for release of radionuclides to well earlier in this chapter, i.e. it is assumed that the toxic metals reach the well at the same rate as the release from the far field, and that the metals reach the well without any delay. For release of toxic metals to well, the concentration in the well is calculated by diluting the annual release with the capacity of the average well on the site in question. The concentration of the metals in the well is compared with relevant guideline values for use of drinking water (SLV, 1993 and USEPA, 1996).

The estimate of the quantity of toxic metals in the different repository parts is presented in Chapter 2 and summarized in Table 8-17. The results for release of toxic metals to well are given in Table 9-3 for Aberg, Beberg and Ceberg at three different times: 100, 1,000 and 10,000 years after closure. The concentrations of lead, beryllium and cadmium obtained are well below assumed comparison values.

Table 9-3 Concentration of toxic metals from SFL 3 and SFL 5 for release to average well in Aberg, Beberg and Ceberg, mg/l.

Time, years	**SFL 3**		**SFL 5**	
	Pb	**Cd**	**Pb**	**Be**
Aberg				
100	$< 10^{-12}$	$< 10^{-11}$	$< 10^{-13}$	$< 10^{-12}$
1,000	10^{-11}	10^{-9}	10^{-13}	10^{-11}
10,000	$5 \cdot 10^{-6}$	$1 \cdot 10^{-4}$	$1 \cdot 10^{-6}$	$1 \cdot 10^{-5}$
Beberg				
100 – 10,000	$< 10^{-13}$	$< 10^{-13}$	$< 10^{-13}$	$< 10^{-14}$
Ceberg				
100 – 10,000	$< 10^{-14}$	$< 10^{-15}$	$< 10^{-14}$	$< 10^{-16}$
Comparison value	0.01 [a]	0.001 [a]	0.01 [a]	0.004 [b]

a) Applicable guideline values for use of drinking water (SLV, 1993)
b) Maximum permissible level in drinking water (USEPA, 1996)

9.5 Design and operation

Some stray materials will get down into the repository during construction and operation of SFL 3-5. Some of these substances will remain after closure. The materials are stray in the sense that they are not present naturally in the environment, nor do they belong to the waste or the structure itself. For SFL 3-5, it is natural to compare the stray materials with what is already contained in the waste, since they are to a large extent the same or related materials.

For the deep repository for spent fuel there is already an estimate of stray materials that are considered likely to enter the repository, first in conjunction with construction and later during deposition of canisters and backfilling. The quantity of stray materials is dependent on the total tunnel length (a larger facility will receive more material). We have assumed that the quantity of left-behind material in SFL 3-5 is 5 times more per metre of tunnel than in the operating galleries in the deep repository for spent fuel (Karlsson *et al.*, 1999). This is due to the fact that the rock debris and dust left from blasting that covers the floor of the tunnel is left behind, and that SFL 3-5 is not washed with a high-pressure spray as thoroughly as the deep repository for spent fuel.

The stray materials consist for the most part of organic materials and various metals and metal oxides. Altogether, the estimated quantity of stray organic materials amounts to about 80 tonnes and metals/metal oxides to 17 tonnes. The metals are relatively harmless chemically (e.g. steel, zinc and aluminium). The organic materials include substances that may be of importance for the long-term performance of the repository. They are surfactants (detergents and degreasing agents, about 1 tonne) and cellulose (sawdust and ventilation dust, about 13 tonnes). The quantities are insignificant compared with what is present in the waste, e.g. 400 tonnes of cellulose, but in SFL 4 and 5, which do not normally have any organic material in the waste (not counting additives in the concrete), the stray materials may be a significant factor.

In the analysis of the reference scenario for SFL 3, the assessment has been made that degradation of all cellulose in the waste to isosaccharinic acid, ISA, cannot give rise to increased release and migration of radionuclides during the first million years. This means that the much lower ISA concentration that might form in SFL 4 and 5 should not influence the release of radionuclides from these repository parts.

9.6 References

Ahlbom K, Äikäs T, Ericsson L O, 1991. SKB/TVO ice age scenario. SKB Technical Report R 91-32, Svensk Kärnbränslehantering AB, Stockholm.

Boulton G S, Wallroth T, Morén L, Kautsky U, 1999. Impact of long-term climate change on a deep geological repository for spent nuclear fuel. SKB Technical Report TR 99-05, Svensk Kärnbränslehantering AB, Stockholm.

Karlsson F, Lindgren M, Skagius K, Wiborgh M, Engkvist I, 1999. Evolution of the geochemical conditions in SFL 3-5. SKB report R-99-15, Svensk Kärnbränslehantering AB, Stockholm.

La Pointe P, Wallmann P, Thomas A och Follin S, 1997. A methodology to estimate earthquake effects on fractures intersecting canister holes. SKB Technical Report TR 97-07, Svensk Kärnbränslehantering AB, Stockholm.

Morén L, Ritchey T, Stenström M, 1998. Scenarier baserade på mänskliga handlingar: Tre arbetsmöten om metod- och säkerhetsanalysfrågor. SKB rapport R-98-54, Svensk Kärnbränslehantering AB, Stockholm.

Nordlinder S, Bergström U, Mathiasson L, 1999. Ecosystem specific dose conversion factors for Aberg, Beberg and Ceberg. SKB Technical Report TR 99-15, Svensk Kärnbränslehantering AB, Stockholm.

Pettersson M, Moreno L, Skagius K, 1999. Analysis of radionuclide migration from SFL 3-5. SKB report R-99-14, Svensk Kärnbränslehantering AB, Stockholm.

SKB, 1999. SR 97 – Main Report. Svensk Kärnbränslehantering AB, Stockholm.

SLV, 1993. Livsmedelverkets kungörelse om dricksvatten. SLV FS 1993:35, Livsmedelsverket.

USEPA ,1996. Drinking water regulations and health advisories. EPA 822-B-96-002. USEPA Office of Water, Washington DC.

10 Discussion and conclusions

10.1 Introduction

This chapter summarizes the assumptions that have underlain the preliminary safety assessment and sheds light on the differences and changes vis-à-vis the previously conducted study (Wiborgh, 1995). Thereupon follows a summary and discussion of the results of the safety assessment, and the uncertainties that might influence the results. Consequences of changes in design are examined, and the chapter concludes with a compilation of the conclusions from the preliminary safety assessment.

10.2 Assumptions

The reference inventory in the preliminary safety assessment is based on a revision of the waste characterization that was done in the prestudy of SFL 3-5. According to present-day forecasts, the waste destined for SFL 3-5 has a volume of about 25,000 m^3. The volume of waste and its total activity content at closure have not changed. The biggest difference is that a greater number of nuclides have been included in the reference inventory. The waste in SFL 5 still makes the biggest contribution to the total activity (more than 80%), and the dominant nuclide is ^{63}Ni. The activity in SFL 3 has increased and the dominant nuclide here as well is ^{63}Ni, since the content of the previously dominant nuclide ^{3}H has declined. The contribution made by SFL 3 to the total activity in the repository is, however, less than 20%. In SFL 4, the activity has fallen slightly and its contribution to the total activity in the repository is less than 0.1%. A compilation of waste volumes, activity contents and radiotoxicity in the different repository parts is given in Table 10-1, see also Chapter 2.

Table 10-1 Information on the waste to SFL 3-5.

Characteristics	SFL 3	SFL 4	SFL 5
Total waste volume	5,600 m^3	10,000 m^3	9,700 m^3
Origin and volume of waste	LILW[a)] from Studsvik, 1,800 m^3 Operational waste from CLAB[b)] and Encapsulation plant, 3,800 m^3	Decommissioning waste from CLAB and Encapsulation plant, 2,200 m^3 Storage canisters from CLAB, 6,500 m^3 Transport casks and transport containers, 1,300 m^3	Core components and reactor internals, BWRs[c)], 7,800 m^3 PWRs[d)], 1,800 m^3 Decommissioning waste from Studsvik, 50 m^3
Packaging	Concrete moulds and steel drums	Steel containers	Long concrete moulds with inner steel cassette
Activity content in 2040	$2 \cdot 10^{16}$ Bq	$7 \cdot 10^{13}$ Bq	$1 \cdot 10^{17}$ Bq
Dominant radionuclides (activity, Bq)	Ni-63, Co-60, Ni-59, C-14, Cs-137	Co-60, Fe-55, Ni-63, Sb-125, Cs-137	Ni-63, Co-60, H-3, Ni-59, Fe-55
Dominant radionuclides, (radiotoxicity, Sv)	Co-60, Ni-63, Am-241, Pu-239, Pu-240	Co-60, Cs-134, Cs-137, Fe-55, Sr-90	Co-60, Ni-63, Fe-55, C-14, Ni-59

[a)] Low- and intermediate-level waste
[b)] Central Interim Storage Facility for Spent Nuclear Fuel
[c)] Boiling Water Reactor
[d)] Pressurized Water Reactor

The layout of SFL 3-5 in the preliminary safety assessment is slightly different from the layout assumed in the prestudy. The layout and design have been made more purpose-suited compared with the preceding version from 1993, see Chapter 3. The foremost change is that SFL 3 and SFL 5 have been given an identical design, allowing greater flexibility. The design is based on experience from construction and operation of the BMA rock vault in SFR 1. The engineered barriers are: the waste packages themselves, surrounding porous concrete, the concrete enclosure and the gravel backfill, see further Chapter 3.

From the viewpoint of the safety assessment, the differences between the new layout and the one in the prestudy are not decisive except in one respect, bentonite is no longer used as backfill in SFL 3. Instead, all spaces are backfilled with gravel. With gravel as backfill instead of impervious bentonite, gas can more easily escape from the enclosure. Gravel instead of bentonite also obviates the necessity of evaluating the stability of the bentonite in the safety assessment. Two factors make such an evaluation difficult in this type of repository. The nature of the waste makes it necessary to consider very long times, and large quantities of concrete are located directly adjacent to the backfill. We must therefore assume that soluble components from the cement can diffuse into the bentonite, and therefore need to analyze the situation in a very long time perspective. Another factor that must be considered if bentonite is chosen as the backfill material is the swelling pressure of the bentonite and the load on the enclosure to which this gives rise. The consequences of different changes in barrier design are discussed in Section 10.6.

Even if bentonite has not been chosen as the backfill material in this study, there are still applications where the sealing properties of bentonite in the repository are very advantageous. One example is the tunnel plugs, another is sealing of ventilation shafts and boreholes. Here as well, the bentonite may come into contact with concrete, but the quantities of concrete are smaller. It is presumably also possible to choose cement types that go well together with bentonite (e.g. low-alkaline cement) or to choose a design that reduces the risk of mutual chemical influence between the different materials.

In the preliminary safety assessment of SFL 3-5, it is assumed that the repository is located on the same site as the deep repository for spent fuel. Freedom of choice as regards siting of the repositories will exist in connection with the future siting process, but here we have chosen the same sites as those included in the SR 97 study: Aberg, Beberg and Ceberg. These sites exhibit differences with regard to chemical groundwater conditions (non-saline and saline waters) and water flows in surrounding rock. The results of the hydrology modelling performed (Chapter 7) serve as a basis for assumed advective travel times for water to the ground surface and chosen discharge areas for the different sites. Assumed repository conditions are presented in Table 10-2. A description of the properties of the geosphere on the assumed sites is provided in Chapter 4, and biosphere conditions are described in Chapter 5. In the previous prestudy of SFL 3-5, only the release of radionuclides from the repository's near field was calculated, and a water flow of 0.1 l/m^2y was assumed to be representative of typical Swedish rock at repository depth.

Table 10-2 Assumed repository conditions in Aberg, Beberg and Ceberg.

Characteristics	Aberg	Beberg	Ceberg
Representative water flows in surrounding rock (l/m^2y)	10	1	0.1
Chemical groundwater conditions	Saline	Non-saline and saline	Non-saline
Advective travel times for water to ground surface (yrs) [a]	10	40	900
Repository depth (m)	300	360	375
Typical ecosystems	Open coast and archipelago	Agricultural land and peatland	Peatland
Capacity of average wells (litres/hour)	300	1,000	500

[a] Theoretical quantity used in the transfer of results from hydro model calculations to transport models.

The potential migration and environmental impact of virtually all radionuclides in the waste, as well as some chemotoxic pollutants, have been analyzed for a reference scenario. The reference scenario describes the expected evolution of the repository's near field when the surrounding far field is stable, i.e. no decisive changes occur in the thermal, hydrological, mechanical and chemical conditions in the rock surrounding the repository, see Chapter 6. Further, we assume that no changes occur in the biosphere, but that today's conditions prevail on the three sites in the future as well.

Besides the reference scenario, a number of other scenarios have been studied, see Chapter 9. Climate change, seismic activity and human actions are examples of events and activities that could influence the future performance and safety of the repository. In this study, however, environmental impact has only been calculated for the scenario with future drilled wells in the vicinity of the repository.

10.3 Summary of results

10.3.1 Biosphere

The reference scenario that serves as a basis for the calculations performed is described in Chapter 6. A detailed description of assumptions and calculations performed for the reference scenario is given in Chapter 8. The calculated environmental impact in the biosphere for the three fictitious sites Aberg, Beberg and Ceberg can be summarized in the following points:

- In Aberg, the highest dose from the release of radionuclides is obtained to the area "Archipelago". The total dose does not exceed $4 \cdot 10^{-3}$ µSv/y. The greatest contribution to the dose is obtained from inorganic ^{14}C in SFL 3 and SFL 5, approximately 10,000 years after repository closure.

- In Beberg, the release of radionuclides to "Agricultural land" gives a maximum total dose of about 3 µSv/y. This occurs about 8,000 years after closure, and the dose is dominated by ^{93}Mo from SFL 5 and SFL 3.

- In Ceberg, the release of radionuclides to "Peatland" gives rise to a maximum total dose of just under 10 µSv/y. This occurs more than 30,000 years after repository closure, and the dose is dominated by ^{36}Cl from SFL 5 and SFL 3.

- On all sites, the releases of the chemotoxic metals lead, beryllium and cadmium give rise to concentrations in the ecosystems that are very low compared with measured natural concentrations.

- A pulse release of organic ^{14}C as gas gives at most a total collective dose of about 0.04 µmanSv locally and regionally, which means that the annual individual dose is much lower.

A comparison between the three hypothetical sites shows that geohydrological conditions and biosphere conditions are vital factors for the calculated impact on the environment. In particular, the ecosystem in the release area is of great importance for the dose obtained from the release of radionuclides from the far field, see Table 10-3. Release to "Coast" or "Archipelago" gives considerably lower doses than release to "Agricultural Land" or "Peatland". There are also differences between "Agricultural Land" and "Peatland". This is most clearly seen in Beberg, where "Agricultural Land" is the primary recipient and "Peatland" is situated immediately adjacent to the discharge area. For the same releases to the ecosystems, "Peatland" gives a greater environmental impact. The dose factor increases for ^{93}Mo, but much more for ^{36}Cl, which means that the latter will be the dose-dominant nuclide for release to “Peatland”.

Table 10-3 Calculated release of dose-dominant radionuclides on the three sites.

Site	Ecosystem	SFL 3		SFL 4		SFL 5	
		nuclide	µSv/y	nuclide	µSv/y	nuclide	µSv/y
Aberg	Archipelago [a)]	C-14	$1 \cdot 10^{-3}$	Sr-90	$3 \cdot 10^{-4}$	H-3	$1 \cdot 10^{-3}$
Beberg	Agricultural Land [b)]	Mo-93	0.6	Mo-93	$<10^{-4}$	Mo-93	2
Beberg	Peatland [c)]	Cl-36	6	Cl-36	$<10^{-4}$	Cl-36	40
Ceberg	Peatland	Cl-36	1	Cl-36	$<10^{-4}$	Cl-36	6

a) Release to Coast $<10^{-4}$ µSv/y
b) Primary recipient
c) Alternative recipient

In addition to the reference scenario, other scenarios have also been studied, see Chapter 9. Except for the reference scenario, environmental impact has only been calculated for the scenario with a future drilled well in the vicinity of the repository. According to the regional hydrology models, no releases will take place from SFL 3-5 in any of the wells that exist today. The calculation therefore assumes that a new well will be sunk in a discharge area from the repository. All releases from the repository are transferred to this well, which is assumed to have a capacity representative of the area. Calculated environmental impact for the well scenario shows that:

- The maximum release of radionuclides from SFL 3-5 to a future well in Beberg is approximately 2 µSv/y and in Ceberg approximately 0.3 µSv/y. The assumed well capacity in Beberg is a factor of 2 greater than in Ceberg, but the releases from the far field to the well in Ceberg are much less. The greatest dose contribution is obtained from the nuclides ^{93}Mo and ^{36}Cl, which derive from the waste in SFL 3 and SFL 5.

- The highest doses are obtained in Aberg, which is the site with the highest groundwater flow and the lowest well capacity (a factor of 3 lower than Beberg). The total dose for release from SFL 3-5 to well is at most about 25 µSv/y. The release from SFL 3 gives a maximum dose of about 6 µSv/y, and release from SFL 5 a maximum dose of about 20 µSv/y. These doses are dominated completely by the release of ^{93}Mo.

- During the first hundred years, when the repository area can be assumed to be protected by institutional controls, the release of ^{3}H and ^{90}Sr could also contribute to the dose from the assumed well in Aberg.

- Releases of the chemotoxic contaminants lead, cadmium and beryllium to an average well never give rise to contaminant concentrations in excess of current standards for drinking water. The highest concentrations are obtained for the average well in Aberg. Of the three sites, the assumed average well in Aberg has the lowest capacity and the least dilution, at the same time as the release from the far field to the well is highest.

10.3.2 Near field and far field

None of the radionuclides that dominate the activity or the radiotoxicity in the waste make dominant contributions to the dose calculated for the three sites. The reason for this is that many of these radionuclides are relatively short-lived and/or are sorbed in the engineered barriers and surrounding rock. They will therefore decay or be retained in the near field and the far field.

Despite the poorer barriers, the dose contribution from the radionuclides in SFL 4 is of subordinate importance. This is mainly due to the lower activity in the waste. The calculated dose is determined by the release of radionuclides from SFL 3 and SFL 5. A detailed account of the release from the individual repository parts from near field and far field is provided in Sections 8.3.3 and 8.4.3.

By plotting the cumulative release of dose-dominant nuclides from near and far fields, an idea can be obtained of how "effective" the near-field and far-field barriers are. This is illustrated by the results of the SFL 3 calculations for the three hypothetical sites. Figures 10-1 and 10-2 shows how large the cumulative release from SFL 3 is as a fraction of the original inventory, for nuclides which dominate the calculated dose.

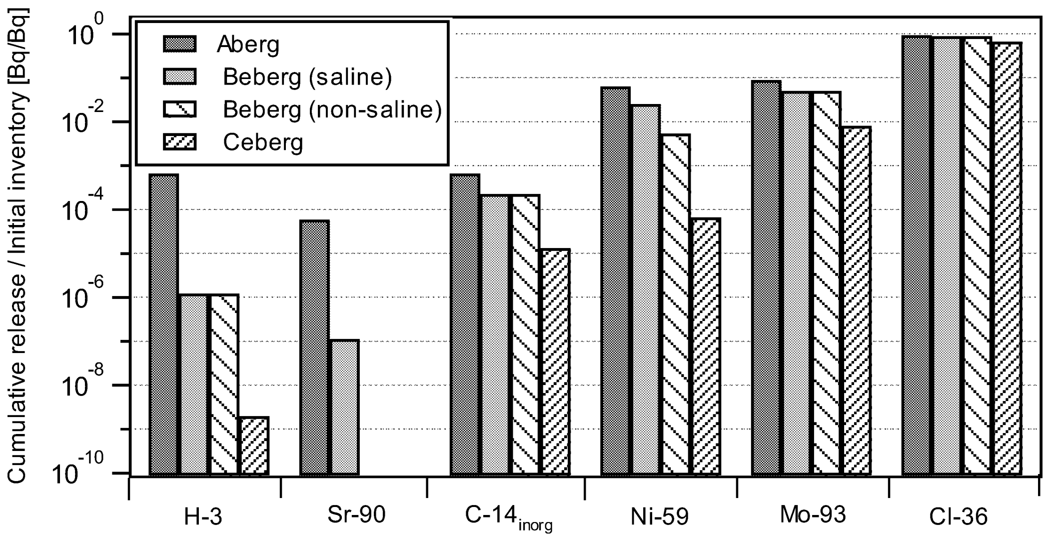

Figure 10-1 Influence of near-field barriers on cumulative release of dose-dominant radionuclides for SFL 3.

The conclusions that can be drawn regarding the importance of the near field are:

- For short-lived nuclides (^{3}H and ^{90}Sr) the near field acts as an important barrier. The release of ^{3}H from the near field in Aberg is 0.1% of the original content in the waste. In Beberg and Ceberg, the release is reduced by a further three and six orders of magnitude, respectively. The tendency is even clearer for ^{90}Sr, which decays almost completely in the near field in Ceberg.

- The near field has very little influence for long-lived and low-sorbing nuclides. All ^{36}Cl comes out in Aberg, Beberg and Ceberg. For ^{93}Mo, with the same K_d but a half-life that is two orders of magnitude shorter than ^{36}Cl, the near field is of some importance for the release.

- The near field is of greater importance for the transport of long-lived and sorbing nuclides (inorganic ^{14}C and ^{59}Ni) than for the transport of long-lived, non-sorbing nuclides, but not to the same extent as for short-lived nuclides. In Aberg and Beberg there is a marked difference in how large a fraction of ^{14}C and ^{59}Ni is released from the near field. This is reasonable, since outward transport in Aberg and Beberg is controlled by diffusion out of the concrete structure, and ^{14}C has a higher K_d in concrete than ^{59}Ni has. In Ceberg, where transport is controlled by the water flow through the backfill, the difference between ^{14}C and ^{59}Ni is not as great.

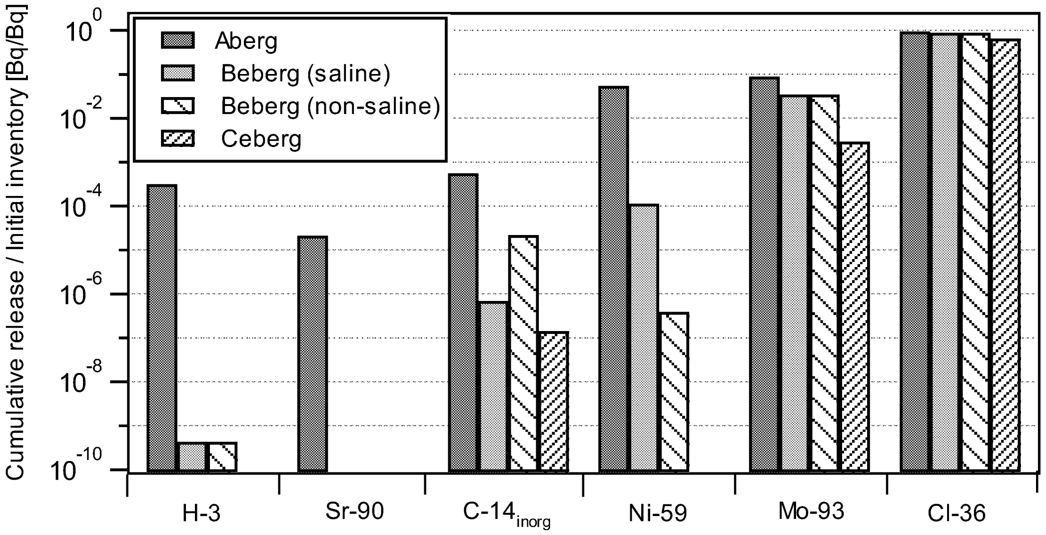

Figure 10-2 Influence of far-field barriers on cumulative release of dose-dominant radionuclides for SFL 3.

The capacity of the far field to act as a barrier to release of radionuclides to the biosphere is controlled by the water's travel time in the geosphere and by the sorbing potential of the nuclides. The latter is determined primarily by the nuclide's sorption capacity (K_d) and the rock's flow-wetted surface area.

The conclusions that can be drawn regarding the importance of the far field are:

- The far field in Aberg is a very limited barrier to almost all nuclides, since the advective travel time for groundwater is very short where the repository has been placed (10 years). The release from the far field is therefore more or less identical to the release from the near field.

- The far field in Beberg reduces the cumulative release by two orders of magnitude or more for short-lived nuclides and for long-lived sorbing nuclides. The far field in Ceberg has an even greater influence. This is especially clear for ^{59}Ni.

- For long-lived nuclides that do not sorb in the rock (^{93}Mo and ^{36}Cl), the far field is of little importance.

10.4 Uncertainties

To reduce the uncertainties in the assessment of environmental impact, an attempt should first be made to reduce the uncertainties inherent in assumptions and calculations for the radionuclides that dominate the dose. These are above all ^{36}Cl and ^{93}Mo. Increased knowledge within the following areas could reduce inherent uncertainties:

- The radionuclide inventory has been arrived at using general correlation factors. Measurements in combination with activity calculations on different metal parts could reduce the uncertainties in the estimates of the inventory.

- These radionuclides occur for the most part as induced activity in metal parts. Uncertainties in the assumed rate at which the metal parts corrode are therefore of importance for nuclide release from the waste. Studies concerning corrosion limitations and other limitations of accessibility could reduce these uncertainties.

- In the concrete barriers, ^{36}Cl and ^{93}Mo are retarded by some sorption, albeit low ($K_d = 0.006$ m^3/kg). There are no equivalent studies for the rock that suggest they might be retarded there. Studies that increase knowledge and reduce uncertainties regarding migration of ^{36}Cl and ^{93}Mo in barrier materials and rock would be of importance.

- The most important uncertainties for the ecosystem-specific dose conversion factors arrived at for different recipients should be analyzed. In the case of ^{36}Cl, consumption of foodstuffs makes a greater contribution to dose than consumption of water. This means that all uncertainties in assumptions and parameters that describe biological processes and exposure pathways can be important.

10.5 Consequences of changes in design

The calculated total dose is dominated by radionuclides in SFL 3 and SFL 5 that have a long half-life and very little or no sorption on the minerals in the rock. The possibility of modifying the design so that the barriers are more effective for these nuclides could be considered. But it is not clear how this could be achieved. Changes for the better in one respect can at worst lead to changes for the worse in another.

A typical dose-dominant radionuclide is ^{36}Cl. It has a half-life of 300,000 years and behaves like a highly mobile chloride anion. To analyze the consequences of modified design, ^{36}Cl is used as a model nuclide.

In the current design, gravel was chosen as a backfill material in SFL 3 and SFL 5. Gravel has a high hydraulic conductivity, so that most of the water that flows through the tunnel flows in the gravel fill. The radionuclides are thereby transported mainly via diffusion through the walls of the concrete enclosure and by advection with the water flowing around the concrete structure, see Figure 10-3. Calculations show that in Aberg, where the water flow is high, the release of ^{36}Cl is controlled by diffusion through the walls of the concrete structure. In Ceberg, on the other hand, where the water flow is low, the release is controlled by the water flow around the enclosure. In Beberg, the release is controlled by both mechanisms (Pettersson *et al.*, 1999).

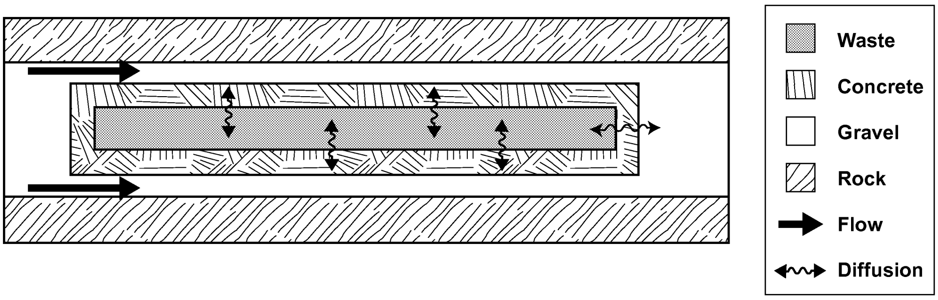

Figure 10-3Schematic illustration of barriers in SFL 3 and SFL 5.

10.5.1 Better concrete enclosure

In Aberg, where diffusion controls the release, the release of radionuclides can be reduced by increasing the enclosure's diffusion resistance. This can be accomplished by increasing the thickness of the enclosure's exterior concrete walls or by reducing the area available for diffusion, for example by lining the walls completely or partially with diffusion-tight material. Increasing the wall thickness from e.g. 0.6 m to 1.2 m would reduce the release of ^{36}Cl by approximately a factor of 2 at a high water flow (20 m^3/y) (Pettersson *et al.*, 1999).

When the flow is low, on the other hand, as in Ceberg, radionuclide release is controlled by the water flow around the enclosure. Improvements of the enclosure leading to an increase in the diffusion resistance therefore have a negligible effect on the release.

In Beberg, increasing the diffusion resistance through the concrete walls of the enclosure reduces the radionuclide release, but the effect is not as striking as in Aberg.

An important prerequisite for these results is that the water flow through the enclosure itself is negligible. This prerequisite is met with a gravel backfill, since gravel has a much higher hydraulic conductivity than the concrete enclosure.

10.5.2 More impervious backfill

An alternative to modifying the concrete structure is to use a backfill material that has a lower hydraulic conductivity than gravel, such as clay. In this case, it is important to take into account the relationship between the hydraulic conductivities of the rock and the enclosure.

If the concrete structure is more conductive than the rock, the total water flow through the tunnel is reduced when the backfill of gravel is replaced with one of clay. The proportion of the total water flow that flows through the concrete structure increases, however, which means that the release of radionuclides increases. If, on the other hand, the conductivity of the concrete structure is equal to or less than that of the rock, a more impervious backfill leads to a lower release.

In other words, it should be ensured that the enclosure is and will remain "better than the rock" before switching to a backfill with low hydraulic conductivity. A more impervious backfill could reduce the release of ^{36}Cl in Aberg up to a factor of ten. A reduction of the release is obtained in Beberg as well, while in Ceberg it is not worthwhile to switch to a more impervious backfill (Pettersson *et al.*, 1999).

10.6 Summarizing discussion and conclusions

At high water flows, such as in Aberg, the barriers in SFL 3 and SFL 5 may need to be improved. The calculations show that the present design may give rise to excessively high releases of long-lived, low-sorbing radionuclides from the near field. Various possible improvements have been discussed. One possibility is to increase the thickness of the concrete structure and thereby increase the diffusion resistance. Big increases are required if the release is to be appreciably reduced. Another way to limit diffusion is to reduce the area through which it takes place. The difficulty lies in finding diffusion-tight materials that are durable over the very long timespans in question. A third option is to backfill the void inside the structure in SFL 3 and SFL 5 with a backfill that is more impervious than a gravel fill, such as clay. The backfill must not be too impervious, however. Gas generated in the repository should preferably be able to escape without building up a pressure that expels water from the enclosure. If a backfill exists that meets these requirements and is durable over very long timespans, it could reduce the release of long-lived radionuclides in Aberg.

Whether or not a site is good enough for the proposed design cannot be decided until after geoscientific investigations and characterizations of the ecosystems on the site. Particularly important are hydrogeological investigations which include measurements in boreholes down to repository depth. The water chemistry, e.g. the salinity of the groundwater, and the suitability of the rock for excavating the rock chambers is naturally also of importance, just to mention some of the factors included in a complete site characterization.

Methods and technology exist for conducting site investigations. Calculations of water flows and radionuclide releases to the possible ecosystems will then show whether the design and the site are feasible, i.e. whether they permit safe disposal of long-lived low- and intermediate-level waste.

The principal conclusions of the completed study are:

- The radionuclides in the waste that are the most important for the evaluation of safety are those that are highly mobile and long-lived. Their being long-lived means that that barriers and the ecosystems must be regarded with a very long time horizon.

- To reduce the uncertainties in the calculation of environmental impact, the uncertainties in the estimates of the dose-dominant radionuclides ^{36}Cl and ^{93}Mo should in particular be reduced. Studies leading to a better understanding of the accessibility of the nuclides in the waste, migration in the barriers and dose impact are also of importance.

- The properties of the site are of importance for safety. Two factors stand out as being particularly important: the water flow at the depth in the rock where the repository is built, and the ecosystem in the areas on the ground surface where releases may take place in the future.

- An unfavourably high water flow in the rock around the repository can be compensated for by better barriers in the near field. However, they must perform satisfactorily over a very long period of time. This requires materials that are durable in the chemical and mechanical environment of the repository.

10.7 References

Pettersson M, Moreno L, Skagius K, 1999. Analysis of radionuclide migration from SFL 3-5. SKB report R-99-14, Svensk Kärnbränslehantering AB, Stockholm.

Wiborgh M, 1995. Prestudy of final disposal of long-lived low and intermediate level waste. SKB Technical Report TR 95-03, Svensk Kärnbränslehantering AB, Stockholm.

List of abbreviations

BWR	Boiling-Water Reactor
CLAB	Central interim storage for spent fuel
DOC	Dissolved Organic Carbon
EDF	Ecosystem specific Dose conversion Factor
HRL	Hard Rock Laboratory
ILW	Intermediate Level Waste
LILW	Low and Intermediate Level Waste
LLW	Low Level Waste
Package (a)	Packaging with its radioactive contents
Packaging (a)	The assembly of components necessary to enclose the radioactive contents completely, e.g. drum or container
PWR	Pressurized-Water Reactor
SFR	Final repository for low- and intermediate-level radioactive waste (at the Forsmark Nuclear Power Station)
SFR 1	Final repository for radioactive operational waste
SFR 3	Final repository for decommissioning waste
SFL 3-5	Deep repository for long-lived low- and intermediate-level waste
SFL 3	Repository part intended for long-lived or toxic waste
SFL 4	Repository part intended for decommissioning waste
SFL 5	Repository part intended for core components and reactor internals

a) IAEA Safety Series No 6 ”Regulations for the Safe Transport of Radioactive Material, 1985 edition”

ISSN 1404-0344
CM Gruppen AB, Bromma, 1999